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Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions

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16. ABSTRACT

This methodology document seeks to provide risk assessors with the guidance necessary to estimate the health risks that result from exposure to toxic pollutants in combustor emissions by pathways other than inhalation. The organization of the document reflects the four-step process of risk assessment (hazard identification, dose-response assessment, exposure assessment and risk characterization). This methodology is not intended to be prescriptive; that is, it does not comprise a set of guidelines or recommended approaches that the U.S. EPA believes should be applied in all circemstances. Rather, it provides a set of procedures that the risk assessor can draw upon, where applicable, to a given assessment. The document describes analytical procedures and computer models that can be used to estimate exposure and risk by a variety of environmental pathways. In addition, it serves as a preliminary source of data for carrying out the risk calculations. The degree of scientific support or uncertainty attendant to each calculation varies widely. Therefore, the appropriate use of these procedures and the discussion of uncertainties surrounding the results remain important responsibilities of the risk assessor.

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Environmental Criteria and Assessment Office
Office of Health and Environmental Assessment
Office of Research and Development
U.S. Environmental Protection Agency
Cincinnati, OH 45268

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This document has been reviewed in accordance with U.S. Environmental Protection Agency policy and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

PREFACE

Incineration is a widely-used method for the treatment or disposal of waste materials such as municipal solid waste, hazardous waste, and sewage sludge. The ability to make careful, scientific estimates of potential risks to human health or the environment from atmospheric emissions is critical to the planning, design and siting of waste combustion facilities.

This methodology document seeks to provide risk assessors with the guidance necessary to estimate the health risks that result from exposure to toxic pollutants in combustor emissions by pathways other than inhalation. Whereas procedures for assessing human health risks from inhalation of pollutant emissions are well established, this methodology enables estimation of the indirect human exposures and health risks that can result from the transfer of emitted pollutants to soil, vegetation and water bodies.

This methodology is not intended to be prescriptive; that is, it does not comprise a set of guidelines or recommended approaches that the U.S. EPA believes should be applied in all circumstances. Rather, it provides a set of procedures that the risk assessor can draw upon where applicable for a specific assessment. The appropriate use of these procedures and the discussion of uncertainties surrounding the results remain important responsibilities of the risk assessor. This document is a revision of an earlier report that underwent review by U.S. EPA's Science Advisory Board and also served as a basis for much of the information presented in the document Assessment of Health Risks Associated with Municipal Waste Combustion Emissions (U.S. EPA, 1987c). Revisions were carried out by Syracuse Research Corporation, under U.S. EPA Contract No. 68-C8-0004, and Oak Ridge National Laboratories' Office of Risk Analysis, under Interagency Agreement DW89932633-01.

Measures planned in the near future to further improve the utility of this methodology include (1) the availability of computer software for the execution of calculations in this methodology and (2) demonstrations of the use of the methodology to test its applicability to specific sites and problems. For more information, please contact Randall Bruins, Environmental Criteria and Assessment Office, U.S. EPA, Cincinnati, OH 45268.

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LIST OF ABBREVIATIONS

A	Area of source (m^2)
Aa	Adsorbed contaminant mass per unit of land in watershed (kg/km^2)
AF	Absorption fraction (%/day)
Ah_i	Area planted to crop i that is harvested (m^2)
a_i and b_i	Empirically determined parameters for each Pasquill atmospheric stability category
A_j	Concentration of pollutant in the jth animal tissue group (μg pollutant/g animal tissue DW)
AWQC	Ambient water quality criteria
B_{Hhk}	Human cancer potency for the hth pollutant by the kth route per(mg/kg)/day
Ba_j	Biotransfer factor for the jth animal tissue group (d/kg)
B(a)P	Benzo(a)pyrene
BCF	Bioconcentration factor (L/g)
BCF_a	Adjusted bioconcentration factor (L/kg)
BCF_u	Unadjusted bioconcentration factor (L/kg)
BD	Soil bulk density (g/cm^3)
BI_{hk}	Background intake for the hth pollutant by the kth route (mg/kg BW/day)
Br_i	Plant-soil bioconcentration factor for the ith plant group (μg pollutant/g plant tissue DW)/(μg pollutant/g soil)
Bv	Air-to-leaf concentration ([μg/g DW]/[μg/g])
Bv_i	Air-to-plant bioconcentration factor for the ith plant group [(μg pollutant/g plant tissue DW)/(μg pollutant/g air)
BW	Body weight (kg)
c	Assumed constant of $1.7x10^{-4}$ (atm-cm)
C	"Cover management" factor (unitless)

LIST OF ABBREVIATIONS (cont.)

CA	Contact amount (mg/cm^2)
Ca_j	Daily human consumption of the j^{th} animal tissue group (g animal tissue DW/kg BW/day)
Cd	Cadmium
Cd_i	Pollutant concentration associated with particles of the i^{th} size category during the portion of the hour in which no precipitation occurs (g/m^3)
Cf	Fish ingestion rate (g/kg BW/day)
$CHEM_a$	Mass of chemical present in the water body in adsorbed form (mg/L)
$CHEM_d$	Mass of chemical present in the water body in dissolved form (mg/L)
$CHEM_{s1}$	Mass of adsorbed chemical per mass of sediment in the water body (μg/L)
$CHEM_{s2}$	Mass of adsorbed chemical per unit volume of water (μg/L)
$CHEM_w$	Mass of dissolved chemical per unit volume of water (μg/L)
Ci_{Lg1}	Contaminant concentration in receiving water for long-term, Tier 1 loading scenario (mg/L)
Ci_{Lg23}	Contaminant concentration in receiving water for long-term, Tier 2/3 loading scenario (mg/L)
Ci_{sh1}	Contaminant concentration in receiving water for short-term, Tier 1 loading scenario (mg/L)
Ci_x	Concentration of contaminant in the receiving water for the short-term or long-term loading scenario (mg/L)
cm	Centimeter
CN	Curve number
Cp_i	Daily dietary consumption of the i^{th} plant group by the appropriate age group (g plant tissue DW/kg BW/day)
C_s	Mean sediment concentration in the water body (kg/m^3)
Cs	Soil ingestion rate (g soil/day)

LIST OF ABBREVIATIONS (cont.)

CT	Contact time (hours/day)
Cv	Concentration of pollutant in air due to volatilization from soil ($\mu g/m^2$)
Cw	Consumption rate of water (L/kg BW/day)
Cw_i	Pollutant concentration in the portion of the hour in which precipitation does occur associated with particles in the i^{th} size category (g/m^3)
Cy	Concentration of pollutant in air due to direct emissions ($\mu g/m^2$)
d	day
D	Particle diameter (μm) (Chapter 3 only)
Da	Diffusion coefficient of pollutant in air (cm^2/s)
Da	Dissolved contaminant mass per unit of land in watershed (kg/km^2) (Chapter 9 only)
DDI	Daily dermal intake (mg/kg BW/day)
d_e	Effective diameter of contaminated area (m)
$DEPd_i$	Total dry deposition for period ($g/m^2/s$)
$DEPw_i$	Total wet deposition for period ($g/m^2/s$)
DI	Daily intake (mg/kg BW/day)
DI_A	Daily intake-adjusted (mg/kg/day)
DI_{hk}	Daily intake for the h^{th} pollutant by the k^{th} route (mg/kg/day)
DIFF	Difference between receptor ground level elevation and plume elevation (m)
Dp	Deposition (g/m^2)
d_p	Distance between plants in a row (mm)
d_r	Distance between rows of plants (mm)
D_R	Depth of runoff in watershed (cm)
D_s	Duration of scenario (years)
DW	Dry weight

LIST OF ABBREVIATIONS (cont.)

Dy	Total yearly deposition rate of pollutant (g/m^2/yr)
Dyd	Yearly dry deposition rate of pollutant (g/m^2/year)
Dyw	Yearly wet deposition rate of pollutant (g/m^2/year)
E_{10}	Annual average PM$_{10}$ emission factor (g/m^2/hr)
EA	Excess air
EDA	Exposure duration adjustment (unitless)
ER	Excess risk (unitless)
ER_h	Excess risk for the hth pollutant for m exposure routes (unitless)
ER_{hk}	Excess risk for the hth pollutant by the kth route (unitless)
ER^{Mix}	Excess risk for a mixture of n chemicals for multiple exposure routes (unitless)
ESP	Electrostatic precipitator
Ev	Average soil evapotranspiration (cm/year)
Fa	Annual mass of contaminant in fallout per unit area (kg/km^2-year)
Fa_j	Fraction of the jth animal tissue group assumed to originate from contaminated soil (unitless)
FBC	Fluidized bed combustor
FDRY	Correction factor for the fraction of the hour in which no precipitation occurs
F_i	Fraction of time of the ith atmospheric stability category (unitless)
Fp_i	Fraction of the ith plant group assumed to originate from contaminated soil (unitless)
Fr	Froude number
F_v	Wind erosion threshold friction velocity (m/s)
Fv	Fraction of pollutant in vapor phase (unitless)

LIST OF ABBREVIATIONS (cont.)

Fw	Fraction of wet deposition that adheres to plant surfaces (unitless)
g	Acceleration of gravity (9.8 m/s^2) (Chapter 3 only)
g	Gram
GIM	Groundwater Infiltration Model
H	Henry's Law constant (pa m^3/mol)
H	Plume height (m) (Chapter 3 only)
h	(subscript) denotes pollutant
ha	Hectare
Hcrit	Critical plume height (m)
HE	Effective height of source emission (m)
HI	Hazard index (unitless)
HI_h	Hazard index for the hth chemical for all routes (unitless)
HI_{hk}	Hazard index for the hth chemical by the kth route (unitless)
HI^{Mix}	Hazard idex for a mixture of n chemicals for multiple exposure routes (unitless)
HMIN	Minimum distance between plume and ground (m)
Hpc	Height of plume above critical height (m)
hr	Hour
Ht	Height of plant above soil (m)
Htc	Height of local terrain above critical height (m)
I	Average annual irrigation (cm/year)
i	(subscript) denotes plant group
Ia	Total daily intake from consumption of animal tissue (μg pollutant/kg/BW/day)
Ia_j	Total daily intake from ingestion of jth animal tissue group (μg pollutant/kg BW/day)

LIST OF ABBREVIATIONS (cont.)

Ip	Total daily intake from consumption of plant tissue (µg pollutant/kg/BW/day)
Ip_i	Human daily intake of pollutant due to the consumption of the ith plant group (µg pollutant/kgBW/day)
I_R	Infiltration rate (cm/yr)
ISCLT	Industrial Source Complex Long-Term Model
ISCST	Industrial Source Complex Short-Term Model
j	(subsrcipt) denotes animal group
K	"Erodability" factor (tons/acre)
k	(subscript) denotes exposure route
k_0	Rate constant for zeroth order losses (year^{-1})
k_1	First-order loss rate (year^{-1})
k_{1D}	First-order loss rate for degradation (year^{-1})
k_{1E}	First-order loss rate for erosion (year^{-1})
k_{1I}	First-order loss rate for infiltration (year^{-1})
K_d	Soil water partitioning coefficient (mL/g) or (m^3/kg)
Ke	Equilibrium coefficient
kg	Kilogram
km	Kilometer
kp	Plant surface loss coefficient (year^{-1})
K_{ow}	Octanol-water coefficient (unitless)
ks	Soil loss constant (year^{-1})
ksg	Loss constant due to degradation (abiotic and biotic)(year^{-1})
ksl	Loss constant due to leaching (year^{-1})
ksv	Loss constant due to volatilization (year^{-1})
Kt	Gas phase mass transfer coefficient (cm/s)

LIST OF ABBREVIATIONS (cont.)

L	Liter
L	Mixing depth of inversion lid (m) (Chapter 3 only)
l	Length of the unit area (mm)
Lc_d	Lipid content of dietary seafood (kg/kg)
Lc_e	Lipid content of experimental organism (kg/kg)
L_d	Lifetime duration (years)
l_f	Length of long fruit (mm)
LOAEL	Lowest-observed-adverse-effect level
LS	"Topographic or slope length" factor (unitless)
m	Meter
MB	Mass burn incinerator
Mg	Megagram
mg	Milligram
MCWI	Maximum cross wind integrated concentration
MI	Modular incinerator
mL	Milliliter
Mm	Maximum contaminant mass per area of soil (kg/km^2)
mol	Mole
M_t	Depth of snow melt for storm event (cm)
MUSLE	Modified Universal Soil Loss Equation
Mw	Maximum contaminant mass per volume of water (kg/km^3)
MWC	Municipal waste combustor
MSW	Municipal solid waste
N	Eddy reflection number (unitless) (Chapter 3 only)
N	Schmidt number for gas phase (unitless)
n	Number of fruit per square meter

LIST OF ABBREVIATIONS (cont.)

NFCS	National Food Consumption Survey
NOAEL	No-observed-adverse-effect level
n_r	Number of plants per row
ng	Nanogram
P	Average annual precipitation (cm/year)
pa	Density of air (1190 g/m^3 at 25°C)
PAH	Polycyclic aromatic hydrocarbons
PCB	Polychlorinated biphenyls
PCDD	polychlorinated dibenzodioxins
PCDF	polychlorinated dibenzofurans
Pd_i	Concentration of pollutant in the i^{th} plant due to direct deposition
P_{ft}	Mass of chemical deposited directly onto the water body (kg)
P_i	Total concentration of pollutant in the i^{th} plant (μg pollutant/g plant tissue DW)
P_{ij}	Total concentration of the i^{th} plant group eaten by the j^{th} animal each day (μg pollutant/g plant tissue DW)
PM	Particulate matter
PM_{10}	Particulate matter ≤ 10 μm
P_{qt}	Mass of chemical entering water body in dissolved form (kg)
Pr_i	Concentration of pollutant in the i^{th} plant due to root uptake (μg/g)
P_s	"Supporting practice" factor (unitless)
Pv_i	Concentration of pollutant in the i^{th} plant due to air-to-plant transfer
P_{xt}	Mass of chemical entering water body in adsorbed form (kg)
P°_L	Liquid-phase vapor pressure

LIST OF ABBREVIATIONS (cont.)

q	Emission rate of the pollutant from the soil ($\mu g/m^2/s$)
Q	Source emission (g/s)
Q	Volume of runoff (km^2-cm) (Chapter 9 only)
q_p	Peak runoff (m^3/sec)
Qp_{ij}	Quantity of the i^{th} plant group eaten by the j^{th} animal each day (kg plant tissue DW/day)
Qs_j	Quantity of soil eaten by the j^{th} animal each day (kg soil/day)
R	Ideal gas constant (L-atm/mole-°K) (Chapter 4 only)
R	"Erosivity" factor ($year^{-1}$) (Chapter 9 only)
R	Radius of the contaminated area (m) (Chapter 5 only)
R_{10}	Annual emission rate of the contaminant
RC	Rotary combustor
Rc	Receptor
RDF	Refuse derived fuel
RE	Relative effectiveness (unitless)
RE_D	Relative effectiveness of the dermal route (reciprocal od the absorption fraction for ingested chemical) (unitless)
R_e	Interception fraction for exposed produce
Ref	Refractory-lined
r_f	Radius of individual fruit or plant (mm)
RfD_i	Reference dose for chronic inhalation exposure (mg/m3)
RfDo	Reference dose for chronic oral exposure (mg/kg/day)
RfD_{hk}	Reference dose for chronic exposure to the h^{th} pollutant by the k^{th} route
R_{lf}	Interception fraction of long fruit (unitless)

LIST OF ABBREVIATIONS (cont.)

R_{lv}	Interception fraction for leafy vegetables (unitless)
R_{mlv}	Interception fraction for mature leafy vegetables (unitless)
r_n	Number of rows of plants
Rpg	Interception fraction for pasture grasses (unitless)
Rp_i	Interception fraction of the edible portion of plant tissue for the i^{th} plant group (unitless)
R_{rf}	Interception fraction of round fruit (unitless)
R_t	Depth of total rainfall for storm event (cm)
R_w	Ratio of wind speed at 7m to friction velocity as a function of roughness height
s	Second
S	Watershed retention parameter (cm)
SA	Exposed skin surface area (cm^2)
SA	Starved air (Chapter 3 only)
Sc	Soil concentration of pollutant (μg pollutant/g soil)
SCS	Soil Conservation Service
S_T	Whitby's average total surface area of particles (cm^2/cm^3)
timef	Time span that analysis will be carried out (years)
T_f	Time frame of the model (years)
T	Temperature (°K)
$t_{½}$	Environmental half-time (days)
Ta	Ambient temperature
TBI	Total background intake of pollutant (mg/kg/day)
Tc	Total time period over which deposition occurs (years)

LIST OF ABBREVIATIONS (cont.)

TCDD	2,3,7,8-Tetrachlorodibenzo-p-dioxin
TDI	Total daily intake (mg/kg/day)
TDI_{hk}	Total daily intake of the h^{th} pollutant by the k^{th} route (mg/kg/day)
TER	Terrain adjustment factor ranging in value from 0.0 to 1.0
TFC	Terrestrial food chain
THT	Height of terrain obstacle above stack base elevation (m)
TOTCHEM	Total mass of chemical in water body (kg)
Tp_i	Length of plants exposure to deposition per harvest of the edible portion of the i^{th} plant group
T_r	Duration of storm event (hours)
[u]	Mean annual wind speed (m/s)
u	Stack top wind speed (m/s)
$u_{6\text{-}hr}$	Maximum 6 hr. mean wind speed (m/s)
U	Mean wind speed at plume height and receptor location (m/s)
USLE	Universal Soil Loss Equation
u_t	Threshold wind speed
V	Fraction of contaminated surface vegetation
V_d	Dry deposition velocity (m/s)
VDF	Vertical distribution function
Vf_x	Appropriate dilution volume for water body under consideration (m^3)
VPTG	Vertical potential temperature gradient (°K/m)
VS_x	Appropriate dilution volume for watershed under consideration over the duration of storm event (m^3)
W	Gravitational settling velocity (m/s)
w	Width of the unit area (mm)

LIST OF ABBREVIATIONS (cont.)

WA_L	Land area of watershed receiving fallout (km^2)
WA_W	Area of water receiving fallout (km^2)
Wc	Water concentration (mg/L)
WW	Water walls
x	Horizontal upwind distance of source from receptor (m)
X_e	Sediment loss rate per unit area watershed over time (kg/km^2-year)
X_i	Concentration of contaminant i in groundwater (mg/L)
X_s	Mass of sediment in the water body (kg)
y	Horizontal crosswind distance from plume centerline to receptor (m)
Yh_i	Harvest yield of i^{th} crop (kg DW)
Yp_i	Yield or standing crop biomass of the edible portion of the i^{th} plant group
yr	year
z	Vertical coordinate of receptor (m) (Chapter 3 only)
Z	Soil depth (m)
σ_{zi}	Gaussian plume dispersion parameter for the vertical direction (m)
σ_y	Gaussian plume dispersion parameter for the horizontal direction (m)
Λ	Scavenging coefficient (1/s)
μ_a	Viscosity of air (g/cm-s)
μg	Microgram
μm	Micrometer
θ	Soil volumetric water content (ml/cm^3)

LIST OF ABBREVIATIONS (cont.)

α	Pollutant concentration in the PM_{10} fraction of soil
α_a	Fraction of chemical adsorbed (unitless)
α_d	Fraction of chemical dissolved (unitless)

1. INTRODUCTION

1.1. PURPOSE

This methodology document seeks to provide risk assessors with the guidance necessary to estimate the health risks that result from exposure to toxic pollutants in combustor emissions by pathways other than inhalation. Procedures for assessing human health risks from inhalation of combustion facility emissions are well established; this methodology sets out procedures for use in estimating the indirect human exposures and health risks that can result from the transfer of emitted pollutants to soil, vegetation and water bodies.

The organization of the document reflects the four-step process of risk assessment (hazard identification, dose-response assessment, exposure assessment and risk characterization) initially described by the National Research Council (NRC, 1983). In this document, however, exposure assessment is presented before the other steps.

This methodology is not intended to be prescriptive; that is, it does not comprise a set of guidelines or recommended approaches that the U.S. EPA believes should be applied in all circumstances. Rather, it provides a set of procedures that the risk assessor can draw upon, where applicable, to a given assessment.

The document describes analytical procedures and computer models that can be used to estimate exposure and risk by a variety of environmental pathways. In addition, it serves as a preliminary source of data for carrying out the risk calculations.

The degree of scientific support or uncertainty attendant to each calculation varies widely, as the careful reader will appreciate. Therefore, the appropriate use of these procedures and the discussion of uncertainties surrounding the results remain important responsibilities of the risk assessor.

1.2. BACKGROUND

Faced with increasing problems of land disposal, many communities are actively considering incineration as a disposal alternative. Incineration is being used as a method of disposal for municipal solid waste (MSW), sludge and hazardous waste. A fundamental issue attendant with the growth of incineration is the environmental impact of pollutants in the combustor emissions. In the past, most analyses of human health risk associated with atmospheric emissions from combustion sources have focused only on exposures occurring by direct inhalation. Recent studies, however, have linked elevated levels of pollutants in soils, lake sediments and cow's milk to atmospheric transport and deposition of pollutants from combustion sources (Fradkin et al., 1988). These studies indicate that deposition of atmospherically emitted pollutants could result in indirect avenues of exposure for humans. Therefore, this methodology was developed to examine indirect exposures and risks to emission from a variety of combustion sources. This methodology does not address exposure and health effects from incinerator ash. A forthcoming document from the

Exposure Assessment Group (U.S. EPA) entitled Exposure Assessment Methodology for Municipal Waste Combustor Residuals Management considers incinerator ash exposure.

One type of incinerator that is being increasingly used is the municipal solid waste combustor (MWC). At least 4% of the greater than 155 million tons of MSW generated each year is incinerated at 111 facilities in the United States. The U.S. EPA projects that by the year 2000, >300 facilities nationwide will incinerate >90 million tons of MSW annually. At present, 210 facilities are in the planning stages or under construction and projected to be complete by 1993. There is a limited data base regarding the type and quantity of pollutants that may be emitted during the combustion of MSW.

Although many MWCs have been tested for compliance with particulate emission standards, a limited number of tests for specific pollutants have been conducted. The U.S. EPA's data base on MWC pollutant emissions consists of test results from three types of MWCs: mass burn, refuse derived fuel (RDF), and modular. Emissions from MWCs are known to contain heavy metals and toxic organic compounds. The metals of concern include arsenic, beryllium, cadmium, chromium, mercury, nickel, and lead. The organics of concern include chlorinated dioxins/furans, polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCB), formaldehyde, chlorobenzenes, and chlorophenols.

Incineration is also a common method of disposal for sewage sludge. The U.S. EPA estimates that approximately 1.7 million dry metric tons of sewage sludge are incinerated each year, accounting for 21% of all sewage sludge disposal (Federal Register, 1989). Currently, 169 facilities use 282 incinerators to dispose of sludge; these facilities represent 1% of the facilities engaged in sludge disposal. The most common type of sludge incinerator is the multiple hearth incinerator; 231 of the incinerators in use are this type. Emissions from the combustion of sewage sludge are likely to contain sulfur dioxide, oxides of nitrogen, heavy metals, toxic organic compounds, and hydrocarbons. The U.S. EPA is proposing numerical limits on the following pollutants for combustion of sewage sludge: arsenic, beryllium, cadmium, chromium, lead, mercury, nickel, and total hydrocarbon. Total hydrocarbon is the sum of all emitted organic compounds. Some specific organics identified in sludge incinerator testing include chlorinated benzenes, benzene, chloroform, dichloroethene, methylene chloride, and trichloroethene.

Incineration is one method available for the disposal of hazardous waste. The most recent data on hazardous waste incineration was obtained from a 1981 U.S. EPA survey (described in Freeman, 1988). In 1981, approximately 265 million metric tons of waste was generated, consisting of mostly nonhazardous products contaminated with smaller amounts of hazardous materials. Approximately 5.5 million metric tons was disposed by thermal destruction (1.7 million metric tons in 240 incineration facilities and 3.8 million metric tons in 1300 industrial boilers and furnaces)(Freeman, 1988).

Of the 240 hazardous waste incinerators operating in 1981, 49% were liquid injection combustors, 19% were fixed hearth combustors, and 6% were rotary kiln combustors. Only 45% of the facilities employed air pollution control technology in 1981. Typical systems included combustion gas quench, venturi scrubber, acid gas adsorber, and mist eliminator (Freeman, 1988).

The major waste streams incinerated in 1981 were spent nonhalogenated solvents and corrosive and reactive wastes contaminated with organics. Hydrocyanic acid and contaminated water were also important waste streams. The typical pollutants emitted from incineration of these wastes include acid gases (HCl, SO_2, NO_x) (Freeman, 1989) and the following products of incomplete combustion: benzene, chloroform, tetrachloroethylene, 1,1,1-trichloroethane, toluene, and methylene chloride (Freeman, 1988, 1989).

1.3. SCOPE

In the past, the U.S. EPA has focused on the health risks posed by inhalation of pollutants in emissions from combustion sources. The Environmental Criteria and Assessment Office (ECAO) of the Office of Health and Environmental Assessment, Office of Research and Development (ORD) has developed this methodology to expand these analytical horizons. This methodology permits the assessment of the potential human health risks posed by indirect exposure pathways resulting from wet and dry deposition and from ambient air concentrations of pollutants emitted from the stacks of combustors. This methodology can be applied to all emitted pollutants (multipollutant).

Pollutants from stack emissions can be incorporated into soil or water and, from there, can accumulate in the terrestrial or aquatic food chains. Humans can be exposed to these pollutants in various media (multimedia) by dermal contact, direct ingestion and/or ingestion of foods grown in the contaminated soil or water.

This methodology guides risk assessors through the four steps of risk assessment to characterize the exposure and risk posed by pollutants in various pathways. The document starts by presenting guidelines on developing site-specific exposure scenarios (Chapter 2). After discussing air dispersion models appropriate for analyzing combustor emissions (Chapter 3), the document describes assessment of exposure through terrestrial and aquatic systems.

For terrestrial systems, the document presents methods for calculating pollutant concentration in soil (Chapter 4) and calculating daily intake through ingestion of food (Chapter 5), ingestion of soil (Chapter 6), and dermal absorption from soil (Chapter 7). Exposure via inhalation of resuspended dust is evaluated (Chapter 8) and found to be an insignificant pathway of exposure to combustor emissions.

For aquatic systems, the document presents methods for calculating pollutant concentration in surface water, rainwater, and groundwater (Chapter 9). Groundwater was found to be an insignificant pathway of exposure to combustor emissions. Methods for calculating daily intake through ingestion of water (Chapter 10), ingestion of fish (Chapter 11), and dermal absorption from water (Chapter 12) are presented. The document discusses hazard identification (Chapter 13) and dose-response assessment (Chapter 14) as they apply to assessment of combustion emissions. Risk characterization is presented in Chapter 15.

The U.S. EPA recognizes that there are uncertainties inherent in the methodology. For example, the methodology can be applied only to the finite number of chemicals that have actually been identified in combustor emissions. There are potentially hundreds of pollutants that may be emitted but have not been looked for or measured. This document uses cadmium and benzo(a)pyrene as examples of how these methods can be applied. Although the most current research available on these chemicals was used in the examples, readers should not consider this document to be a definitive statement of exposures and risks posed by these two chemicals. U.S. EPA (1988a) has prepared a document similar to this methodology that focuses on exposures to and risks posed by dioxins present in combustor emissions.

The document uses emission rates that were, for the most part, measured during good operations of the incinerator. Therefore, emissions resulting from perturbations, mechanical failures, excessively wet garbage feed, shutdown and cold start of the plant are not reflected in the exposures or risks presented.

There are other data limitations that ultimately constrain the methodology. This document uses the most current national average values as input into the example calculations for the purpose of illustrating the methods. Readers should consider that many of the factors affecting exposure vary according to region, climate, or local habits. Risk assessors should use local data to conduct a site-specific assessment whenever possible to reduce the uncertainty associated with applying national data to a local community.

The methodology, therefore, is not a comprehensive environmental audit, but is best regarded as an evolving and emerging process that moves the U.S. EPA beyond the analysis of potential effects on only one medium (air) and exposure pathway (inhalation) to the consideration of other media and exposure pathways.

2. HUMAN EXPOSURE SCENARIOS

2.1. INTRODUCTION

Pollutants emitted to the atmosphere from stationary combustion facilities may be deposited on environmental media, i.e., soil, water and vegetation, near the combustor. Humans or other organisms can be exposed to the emitted pollutants by various pathways such as dermal contact with the pollutants in soil or water, consumption of the exposed environmental media, consumption of animals that have ingested exposed soil, water or vegetation, or consumption of fish living in polluted waters. For humans, an important part of a multimedia, multipollutant exposure assessment is defining the individual and/or population in the vicinity of the combustor and the potential pathway(s) by which the individual(s) may be exposed to the emitted pollutants.*

Exposure scenarios are real or hypothetical situations that define the source, individual(s), the pathway(s) of exposure and the variables that affect the exposure pathways. Once the exposure scenarios are defined, quantitative exposure and risk estimates for adverse health effects may be developed.

This chapter presents guidelines for developing site-specific exposure scenarios. Actual exposure scenarios can be constructed on a site-specific basis by the exposure/risk assessor using the various pathways presented in this document (Chapters 4-12). Three hypothetical exposure scenarios will be developed in this chapter to illustrate possible approaches. These three scenarios will be used throughout the document to illustrate calculations for the exposure pathways. Scenario "A" will be constructed to represent the most likely occurrence for an exposed population anywhere within a 50 km radius of the facility, and typical or average values of input variables are chosen. In contrast, Scenario "C" will illustrate the hypothetical case wherein all variables are maximized so that the highest potential exposure to an individual is determined. While most persons in the vicinity of a combustion facility are not likely to have such an exposure, this possibility may exist and should be explored. Scenario "B" will strike an approximate midpoint between Scenarios A and C; exposures will be higher than those in Scenario A but not as infrequent as those in Scenario C. These scenarios will be referred to in the following chapters and more fully developed in Section 2.6. The scenarios constructed in this document are examples only and the risk assessor can define/construct exposure scenarios for any site. Uncertainties in the variables that affect the exposure pathways will be delineated in the chapter for each pathway.

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*This methodology is geared toward protection of the general public and the environment. Occupational exposures are not considered.

2.2. CREATING EXPOSURE SCENARIOS

Contaminants associated with atmospheric emissions from combustion sources may be deposited on the surfaces of soil, vegetation and surface water bodies downwind from the source. Human exposure scenarios can be developed by defining the individuals that could be exposed; the characteristics of the source, such as its length of operation; the environmental media in the surrounding area; and the possible routes of exposure to the contaminated media. Numerous exposure scenarios are, therefore, possible. Figure 2-1 presents a decision tree that can be used to define the exposed individual(s) and select appropriate exposure pathways to tailor a site-specific assessment.

The exposed individual(s) must first be defined in terms of age, location relative to the combustion source, and length of time spent at that location. Although the age of the individual will affect exposure routes and pathways, defining age in general terms such as child or adult generally suffices for exposure assessment. In most cases, location will refer to the residence of the individual. The length of time (i.e., number of years) that the individual spends at the defined location influences potential pathways of exposure and exposure levels.

Various characteristics of the combustor affect human exposure. These factors include the type and capacity of the combustor, emission rate, emission control technology such as electrostatic precipitators, the type of materials incinerated, and the length of time the combustor has been in operation. These characteristics will be further discussed in Chapter 3.

The environmental media likely to be exposed to deposited emissions should be defined, since the environment surrounding the source also contributes to selection of the pathways by which the individual(s) may be exposed to the emitted pollutants. Potentially contaminated environmental media include food commodities, soil and water. The potential routes of exposure to

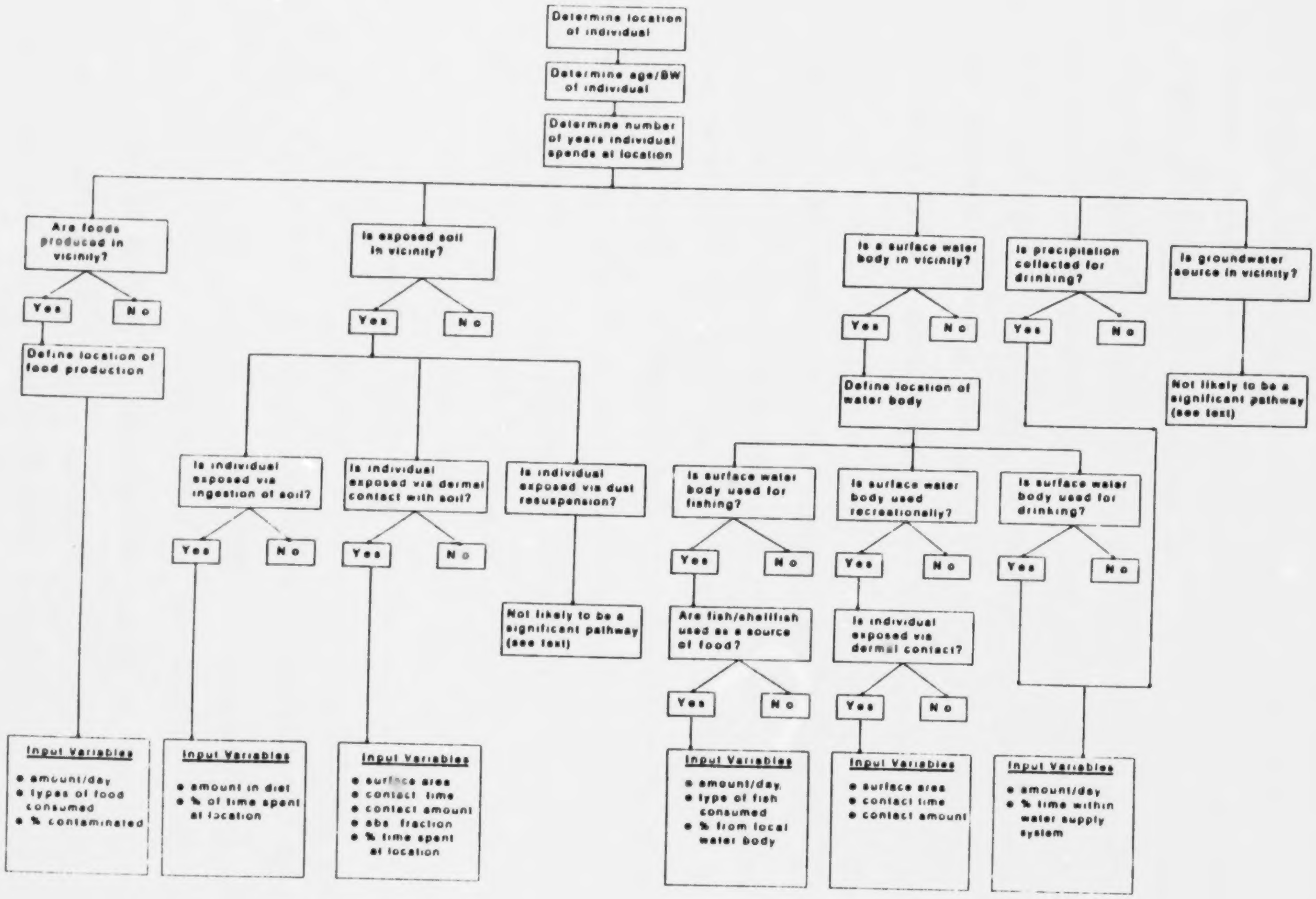
Determine location of individual
Determine age/BW of individual
Determine number of years individual spends at location
Are foods produced in vicinity?
Yes
No
Define location of food production
Input Variables
• amount/day
• types of food consumed
• % contaminated
Is exposed soil in vicinity?
Yes
No
Is individual exposed via ingestion of soil?
Yes
No
Input Variables
• amount in diet
• % of time spent at location
Is individual exposed via dermal contact with soil?
Yes
No
Input Variables
• surface area
• contact time
• contact amount
• abs. fraction
• % time spent at location
Is individual exposed via dust resuspension?
Not likely to be a significant pathway (see text)
Is a surface water body in vicinity?
Yes
No
Define location of water body
Is surface water body used for fishing?
Yes
No
Are fish/shellfish used as a source of food?
Yes
No
Input Variables
• amount/day
• type of fish consumed
• % from local water body
Is surface water body used recreationally?
Yes
No
Is individual exposed via dermal contact?
Yes
No
Input Variables
• surface area
• contact time
• contact amount
Is surface water body used for drinking?
Yes
No
Input Variables
• amount/day
• % time within water supply system
Is precipitation collected for drinking?
Yes
No
Is groundwater source in vicinity?
Not likely to be a significant pathway (see text)

Figure 2-1. Decision Tree for Defining Exposure Scenarios

the contaminated media include ingestion, dermal absorption and inhalation. The topography of the area and its meteorologic and climatic conditions influence determination of relevant pathways (also discussed in Chapter 3). For example, consumption of water and fish may become pathways for exposure to the emitted pollutants if a surface water body, such as a lake or stream, are located in the area surrounding the combustion facility. Whether the surrounding countryside is farmland or an urban area influences the importance of the soil pathways in constructing a site-specific exposure scenario.

Once the individual and environment are defined and the potential pathways of exposure chosen, mathematical models of environmental fate and transport can be used to estimate the concentration of pollutants in the various environmental media in the vicinity of the combustor. In this document, "vicinity" is used to represent the area within a 50 km radius of the source. The remaining chapters present models used to estimate pollutant concentration in food, soil and water, and to calculate daily intake of pollutants by the various routes of exposure. By varying the values of data entered into the models from typical to worst-case, a range of potential exposures can be estimated. For example, in the Terrestrial Food Chain Model, the amount of contaminated food consumed may be varied from a minor fraction to most of the food source being contaminated in order to determine a range of exposure for that pathway. The risk posed by each exposure pathway is assessed by comparing the estimated exposures with health-based criteria. The exposure pathways are summarized in Section 2.4. and described in detail in Chapters 4-12. Estimation of human health risks is delineated in Chapters 13-15.

2.3. DEFINING THE LENGTH OF TIME OF EMISSIONS

In this methodology, individuals are assumed to be exposed to long-term average soil or water levels of the pollutant present in the environment following a period of time in which there has been continuous combustor emissions. Neither upset conditions nor intermittent emissions are considered in the present methodology. Varying lengths of time of operation of the combustor can be used to estimate the pollutant concentration. The use of combustion facility characteristics and local topography and meteorology to model contaminant deposition rates is described in Chapter 3. Determination of soil and water concentrations resulting from deposition are described in Chapters 4 and 9, respectively. For example, in the case of a municipal waste combustor (MWC), the lifetime of the facility (or total period of deposition) could be considered to be approximately 30 years. Since the site is already dedicated for a MWC, it could further be assumed that the MWC will be replaced and the lifetime of combustor emissions could be as great as 100 years. In this document, 30, 60 and 100 years of continual fallout from a combustion facility will be used to construct Scenarios A, B and C, respectively.

2.4. DEFINING THE EXPOSED INDIVIDUAL

2.4.1. Location of the Individual Relative to the Combustion Source. The ground-level concentration and deposition rate of pollutants emitted from the combustion source are affected by air dispersion of pollutants. Numerous factors such as source characteristics (e.g., stack height), topography, meteorological data (e.g., wind speed and direction) and distance from the source determine the air dispersion and resultant deposition of the emitted pollutants. Annual ground-level concentration of emitted pollutants decreases with distance from the source as shown in Figure 2-2 and Table 2-1, which are based on a modeling analysis of two sewage sludge combustors. Maximal ground-level concentrations are usually measured at distances in very close proximity to the source, although local meteorological conditions and source characteristics play a role. For the combustors used in Figure 2-2, maximal deposition is approximately 0.2 km from the facility. At 2 km, the ground-level concentration has attenuated to approximately 4-14%, and at 50 km from the source the concentration is <0.02% of the maximum ground-level concentration. In generalizing this information to other combustors, the location of the individual relative to the combustor becomes an important determinant of exposure. Generally, individuals located closer to the source would be expected to have higher direct exposure to emitted pollutants than those at further distances from the source. In most cases location will refer to the residence of the individual. A range of exposures for individuals representing the

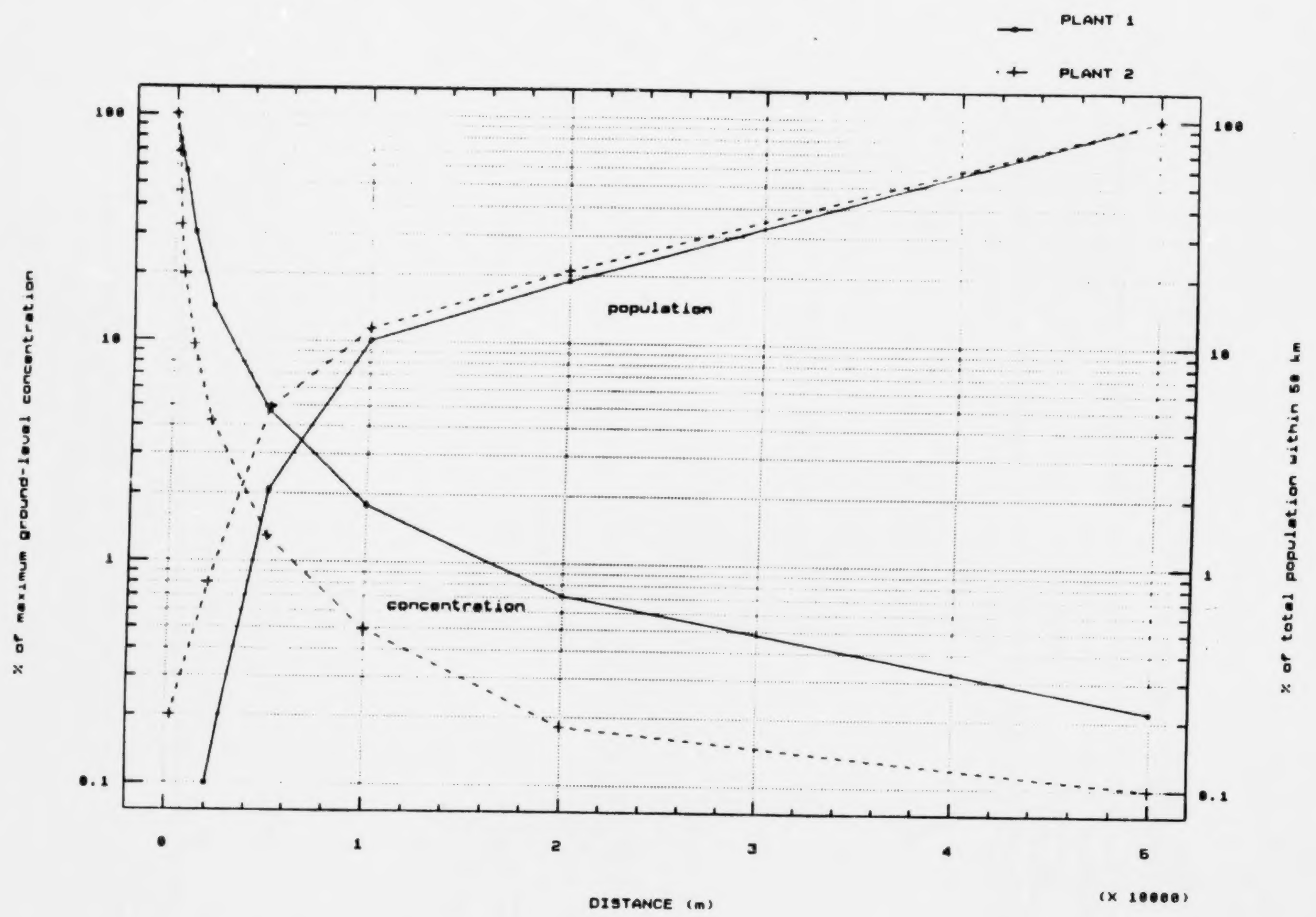

Figure 2-2. Cumulative population (expressed as percentage of the total population within 50 km) and ground-level concentration (expressed as percentage of maximum ground-level concentration) as a function of distance from the facility where ground-level concentration is the areal average within each distance.

TABLE 2-1

Ground Level Concentration of Pollutants and Population Size Surrounding Two Model Combustors

	Distance from the Plant (km)[a]	Relative Population Size (%)[b]	Relative Concentration (%)[c]
Model Plant 1	50	4.0×10^6 (100%)	0.2
	10	4.1×10^5 (10%)	1.8
	5	8.3×10^4 (2.1%)	4.7
	2	2.3×10^3 (0.06%)	14
	0.2	$<2.3 \times 10^3$ (<0.06%)	71
	0[d]	(<0.06%)	100
Model Plant 2	50	3.7×10^5 (100%)	0.04
	10	4.2×10^4 (11%)	0.5
	5	1.8×10^4 (4.9%)	1.3
	2	3.1×10^3 (0.8%)	4.2
	0.2	6.8×10^2 (<0.2%)	45
	0[d]	(<0.2%)	100

[a] Distance refers to radial distance from the source

[b] Population within area of specified distance and equivalent percent of population relative to total population within 50 km radius of the facility

[c] Percent of maximum ground-level concentration at specified distance, where ground-level concentration is the areal average within each distance

[d] This distance refers to the source location

general public could be estimated by choosing several distances from the combustor. In this document, different distances from the facility will be used for exposure Scenarios A, B and C.

Another factor that varies with distance from the combustion facility is the size of the population. Figure 2-2 shows that cumulative population increases as distance from the combustion facility increases; increasing radial distances from the source encompass more of the population. In this document, a 50 km radius around the combustion facility will be used to define the vicinity around the source. The total population in this area then represents all persons who could be exposed to the emissions, excluding those persons that temporarily travel into the area (e.g., for work). At a 10 km radius of the facility, approximately 10% of the total population is found; within a 5 km radius, 2-5% of the population resides. Therefore, the individual used in each of the three exposure scenarios in this document can also be described in terms of the size of the population he represents. For example, an individual in Scenario C resides in close proximity to the combustion facility and represents a small group of people with high exposure because concentration of the emitted pollutants are maximal and have not significantly attenuated over the short distance. An individual residing at a 50 km distance from the source will be exposed to low levels of the emitted pollutants and, therefore, will be more representative of the general population within the vicinity of the combustor.

In this document, a person residing at the point of maximum deposition, a 0.2 km radius from source, is chosen for Scenario C.

As shown in Figure 2-2, this individual will come from a group of <0.3% of the total population who reside at a point where the areally averaged ground-level concentrations are approximately 75% of maximum ground-level concentrations. Although this is a hypothetical situation, it is not unrealistic in that there may be some areas in which a combustion facility is "across the street" or "a block away" from a residential area. The exposure Scenario B will be represented by an individual residing at 5 km and exposed to the average concentration of pollutants within that 5 km radius of the source. This individual represents 2-5% of the total population. Scenario A is that of an individual living 50 km from the facility, exposed to the areally-averaged ground-level concentration of pollutants within that 50 km radius of the source. Determination of the ground level concentrations for these scenarios is presented in Chapter 3.

2.4.2. Age of Exposed Individual. The age of the individual (child vs. adult) influences exposure to the emitted pollutants since pathways, duration and quantities of exposure may vary with age. The daily activities of the individual, amounts of food and water consumed, types of food consumed and exposed skin surface area differ for children and adults. Health-based criteria are also different for children and adults. Therefore, for some exposure pathways such as soil ingestion, children may have a greater exposure and be at greater risk than adults. Human lifespan is generally considered to be 70 years (U.S. EPA, 1986a), and childhood represents approximately 10% of the lifespan, or 6 years. In actual exposure scenarios, individuals may be exposed only during childhood or adulthood. In other cases, exposure may overlap these periods, such as in the case where a child grows into adulthood and remains in the same geographical area. For the three hypothetical exposure scenarios used in this document, a child will represent the individual in Scenarios B and C; an adult will represent the individual in Scenario A.

Childhood is defined in the literature differently by various authors. Data for input into the exposure pathways described in Chapters 4-12 may not be available for children <7 years, or may be presented in the literature for more restrictive age groups such as 2 year olds or children 4-6 years old. The definition of childhood is, therefore, constrained by available data and may vary somewhat for the following exposure pathways. Interpolations or extrapolations of the available data are sometimes made to estimate values for children 1-7 years old.

2.4.3. Human Body Weight (BW). The choice of body weight for use in risk assessment depends on the definition of the individual at risk, and that in turn depends on exposure and susceptibility to adverse effects. In this document the individual at risk is defined in each exposure scenario. The body weight used to determine daily intake is that of an adult or a child, or both, depending on the scenario. Scenario C is a child who grows to an adult and is exposed for his 70 year lifetime. In Scenario B, the individual is exposed for a total of 30 years (as a child and then for part of his adulthood). In Scenario A, an adult is exposed for 16 years.

The daily intake for a pathway was defined as the dose rate on a body-weight basis. Ingestion exposures on a body weight basis are substantially higher for infants and toddlers than for teenagers or adults. Certain behaviors, such as mouthing of dirty objects or direct ingestion of soil, which could also contribute to exposure, are also much more prevalent in children than adults. Therefore, infants and toddlers may be at greater risk when exposure is by food or soil ingestion. The effects, however, may have a long latency period, in some instances approaching the human lifespan. In these cases it may be reasonable to estimate daily intake using the adult values of body weight. In cases where effects have a shorter latency (<10 years) and where children are known to be at special risk, it may be more appropriate to use a body weight for toddlers or infants or for young children. For chemicals that are classified as carcinogens, adult body weight is usually assumed to derive unit risk estimates for air or water. If exposure is lifelong, values of body weight are usually chosen so as to be representative of adults. In this document, 70 kg is used as the body weight for adults. The average body weight for children 1-7 years old is 17 kg (Nelson et al., 1969). This value is used for exposure pathways where information for the inputs in the pathway are available for children 1-7 years old. In some instances, information is not available for all children in this age range and the body weight for children is constrained by the form of the data. For example, the food ingestion rate for meat groups is presented only for 2 year olds (U.S. EPA, 1989b). The body weight chosen, then, to determine daily intake (mg/kg/day) via food ingestion is the mean body weight of male and female 2 year olds (Nelson et al., 1969). The choice of body weight is made to coincide with the available data, so that the daily intake (mg/kg/day) is accurately determined. The form of the input data and body weight value used for determination of daily intake (DI) are delineated in the chapter for each pathway.

2.4.4. Length of Time Individual Spends at Location. A person may possibly have a lifetime exposure to emissions from a combustion source; however, census data on population mobility suggest that many Americans do not remain in the same area for their 70-year lifetime (U.S. Bureau of the Census, 1986). Therefore, a shorter period of time may also be considered to estimate shorter duration exposure to combustor emissions.

An estimate of the number of years a person is likely to spend in a given area, such as in the vicinity of a combustion facility, can be derived from information about mobility rate and median time in a residence. Mobility rate refers to the number of persons over 1 year of age (i.e., percent of the U.S. population) moving to a different residence in the United States within a given year. Data derived from the U.S. Bureau of the Census (1986) show that the mobility rate in 1984 was 17% of the total population. Local moves, those within the same county, accounted for 60% of all moves. It could be inferred from this information that the majority of those that move would remain in the vicinity of the combustion source. However, the likelihood that these short distance moves will influence exposure based on factors such as atmospheric transport of pollutants can not accurately be predicted. Overall rates and volume of geographical movement (number of movers) vary as a function of age, region of the country and type of settlement (i.e., metropolitan area). Mobility rates have tended to decrease with increasing age as shown in Table 2-2. However, since mobility of the U.S. population may change over time, it cannot necessarily be inferred that these rates will remain constant in the future, e.g., over the next few decades. For example, the rates decrease in farming areas; once a farm is established, its owners tend to stay for many years.

The average number of years a person stays in his/her dwelling also affects exposure. This median length of residence in 1980 was 30 years for homeowners, 4 years for renters, and 16 years for owners and renters combined (Burke, 1986).

In addition to the number of years at a particular location or residence, the amount of time spent at that location each day directly affects exposure. For example, children that attend day care or adults that work in a different location for part of the day may be exposed to higher or lower contaminant levels. This issue is further discussed under the relevant pathways in Chapters 4-12.

This methodology examines a range of exposures by varying the number of years spent in the vicinity of the combustor. In

Table 2-2

Geographic Mobility Rates as Related to Age*

Age (years)	Mobility Rate
1-4	26%
5-19	17%
20-29	33%
30-44	17%
45-64	8%
≥65	5%

*Source: U.S. Bureau of the Census, 1986.

Scenario C, the individual is assumed to have spent an entire 70-year lifetime in the vicinity of the combustor. In exposure Scenario B, the individual is assumed to have lived 30 years in the vicinity of the combustor based on the median length of residence for homeowners. In Scenario A, the individual is assumed to have lived 16 years near the combustor based on the median residence time for owners and renters. For a site-specific assessment, local data should be used, if available.

2.5. PATHWAYS OF HUMAN EXPOSURE

As explained earlier, an exposure pathway consists of exposed environmental media and the routes by which individuals contact these contaminated media. Figure 2-1 can serve as a guide to choosing which pathways are relevant for a particular site-specific exposure assessment. In pathways involving soil or water, different routes of exposure are possible and the different ways in which the media are used affect human exposure. Additionally, for the food and surface water pathways, the location of these media relative to the combustor should be defined since these could be different from that of the combustor.

2.5.1. Food Ingestion Pathways. Plants and animals near the stationary combustion source may take up emitted contaminants from the air and or deposited contaminants from the soil. Humans are exposed to the contaminants via the food chain when these plants and animals are produced as a food source.

2.5.1.1. LOCATION OF FOOD PRODUCTION -- Location of food production relative to the combustion source is an important factor in human exposure to emitted pollutants via the food chain. Some of an individual's diet may come from food sources (e.g., gardens, farmland, grazing cattle) within the vicinity (i.e., within a 50 km radius of the emission source), whereas other foods may be imported from distant areas.

The location of the food production, i.e., distance from the source, can be varied to determine a range of potential exposure. Assuming that all of a person's diet comes from produce grown at the site of maximal deposition, whether that be a home garden or farm, and assuming that all meat and dairy products come from animals grazing at the maximum point of emission impact will predict very high exposures. Other scenarios can be constructed by assuming all foods or certain food groups (e.g., milk and dairy) come from the areas of lesser fallout. Section 2.5.1.2 further discusses these scenarios.

2.5.1.2. INPUT VARIABLES FOR EXPOSURE VIA FOOD -- Determination of human intake of contaminants from food is based on the types of foods consumed and the amount of food consumed per day, the concentration of contaminant in the food and the percentage of the diet contaminated (see Figure 2-1).

The types of foods consumed will affect exposure because different plants and animal tissues will take up contaminants at different rates. Therefore, the concentration of the contaminant in the food is determined, in part, by the type of food and varies with the types of food in the diet. The types of foods consumed vary with age, geographical region and sociocultural factors. These issues as well as specific values for various types of foods consumed are discussed in Chapter 5.

The amount of daily food consumption varies according to age, sex, body weight, and geographic region and varies within these categories as well. Values for consumption rates are available from USDA food consumption surveys and are discussed in more detail in Chapter 5. In this document, food consumption rates from the 1977-1978 USDA Food Consumption Survey are used. More recent data (1987-1988), when available, should be used. If, however, in a site-specific assessment, there is more agricultural practice in the area, the USDA 1966-1967 Food Consumption Survey may be more appropriate, since the more recent rates represent mean values of a more suburban lifestyle and may not accurately represent an agrarian population.

The percentage of locally grown (i.e., within the 50 km radius of the combustor facility) food consumed by the individual will affect exposure; all of the dietary intake may not be contaminated. The proportion of local food production will depend, in part, on the degree of urbanization. Local food production can refer to either a home garden or commercial farm. People living in rural or suburban areas who can raise animals and gardens will have a larger percentage of their food produced locally than people living in a city. In this document, only food grown near the combustor is assumed to be contaminated and only a proportion of the individual's diet will be produced locally.

2.5.2. Soil Exposure Pathways. As illustrated in Figure 2-1, if there is soil exposed to deposited emissions in the area surrounding the combustor, soil ingestion, dermal exposure to soil and inhalation of resuspended dust are potential pathways of exposure. Contaminant concentration in soil will vary with distance from the source since deposition rate is related to distance. It is assumed in this document, that soil at the individual's location is contaminated.

2.5.2.1. USE OF THE SOIL -- An individual could be exposed to contaminants in soil by ingesting soil, absorbing the contaminant in soil through the skin or inhaling resuspended dust. Potential routes of exposure are determined by the way the soil is used. Soil that is used for farming or recreation will be involved in pathways for human exposure that are different from soil on roadways or in urban areas. For urban populations, dermal contact and dust resuspension may be more important pathways of exposure than soil ingestion; whereas, for children playing outdoors in suburban or rural areas, soil ingestion and dermal contact may be the more significant routes of exposure.

2.5.2.2. INPUT VARIABLES FOR SOIL INGESTION -- Children and adults are directly exposed to contaminants in soil when they consume soil that may be adhering to their hands. In addition, young children may intentionally eat soil (i.e., a condition known as pica). The factors that affect exposure from soil ingestion include soil concentration, the rate of soil ingestion during time of exposure and the length of time spent in the vicinity of contaminated soil. Soil ingestion rates in children are based on studies that measure the quantities of nonabsorbable tracer minerals in the feces of young children. Ingestion rates for adults are based on assumptions about exposed surface area and frequency of hand to mouth contact. Both indoor dust and outdoor soil may contribute to total daily ingestion. Specific values for ingestion rates are presented in Chapter 6. The amount of time the individual spends in the vicinity of soil exposed to deposition of emitted pollutants (both indoors and outdoors) also influences exposure levels. The worst-case scenario for soil ingestion is undoubtedly a child with pica in close proximity to the combustor. An average scenario may be an adult with a small home garden.

2.5.2.3. INPUT VARIABLES FOR DERMAL EXPOSURE TO SOIL -- Humans will be exposed to contaminants by absorption through the skin when they come in contact with contaminated soil. Surface area, contact time, contact amount, amount of time spent near the combustion source and fraction of contaminants absorbed through skin are some of the factors that affect dermal exposure.

In general, the larger the surface area, the more contaminant that can be absorbed through the skin. Surface area is affected by age and body weight; children have less total surface area than adults. The amount of surface area available for exposure to soil is also affected by the amount of clothing worn. An adult working in the garden in long sleeves and pants will have a smaller exposed surface than an adult working in shorts and a short-sleeved shirt.

For dermal exposure from soil, the exposed surface area dictates the contact amount, the amount of soil that can adhere to exposed skin.

The longer contaminated soil stays in contact with the skin, the greater the amount of contaminant that can be absorbed. Contact time refers to the duration of time each day that contact with soil is possible. The amount of time each day spent in the vicinity of the combustion source where soil is likely to be exposed to emitted pollutants also affects dermal exposure. Both indoor dust and outdoor soil may contribute to daily contact hours. Seasonal exposure can also be considered since regional climate will influence this variable.

The amount of contaminant that can be absorbed through the skin (i.e., absorption fraction) depends on the chemical properties of the contaminant, properties of the soil matrix, and dermal pharmacokinetics. If a compound cannot readily be absorbed through the skin, the daily intake of the compound may be small even if other exposure characteristics, such as contact time, are favorable. Specific values for these variables are discussed in Chapter 7.

2.5.2.4. INPUT VARIABLES FOR EXPOSURE BY RESUSPENSION OF DUST -- Pollutants in the contaminated soil can be resuspended as particulates in the air by wind erosion. The amount resuspended is dependent on the moisture content of the soil, the fraction of vegetation cover, the wind velocity, soil particle size, the pollutant concentration in the soil and the size of the contaminated area. As a consequence to resuspension of the dust, persons may inhale the pollutant particles.

Methodologies have been developed to assess the exposure to pollutants resuspended by wind erosion for landfills (U.S. EPA, 1985a) and Superfund sites (U.S. EPA, 1988b). Application of these methodologies to the deposited combustor emissions indicates that dust resuspension by wind erosion is not a significant pathway. This is further discussed in Chapter 8.

2.5.3. Water Exposure Pathways. The water exposure pathways can be used to determine the concentration of emitted contaminants in groundwater, surface water bodies, or collected precipitation, such as in a cistern. Estimation of daily exposure of individuals using these water sources for various purposes, such as fishing, swimming and drinking water, is possible using various models. Possible routes of human exposure to contaminants via water are shown in Figure 2-1.

2.5.3.1. LOCATION -- Contaminants emitted from the combustor are subject to dissolution from precipitation events and then either move over the ground into a surface water body or infiltrate into the ground and recharge the groundwater. Since annual ground-level of contaminants decreases with distance from the source (see Section 2.3.1), the location of the precipitation collection apparatus, surface water body on which emitted pollutants are deposited, and the soil concentration (affects leachate concentration) are important determinants of the water concentration. Additionally, the location and size of the watershed affect the concentration of contaminants suspended in runoff for the Surface Water Model.

2.5.3.2. USE OF THE WATER -- The way in which the water is used, whether it be collected precipitation or groundwater, or a surface water body such as a lake, farm pond or city reservoir, will affect possible exposure routes. Use of the surface water body for recreation or as a drinking water source will introduce dermal contact and water ingestion as possible exposure pathways. Commercial and/or recreational fishing with subsequent use of the fish and shellfish as a food source, make the food chain an important route of exposure for communities with a surface water body in the vicinity of a combustion source. Ingestion of the water is an important route of exposure if the surface water body is a reservoir; whereas, if the surface water body is used recreationally for swimming and boating, exposure to pollutants by dermal contact with the water is possible. The relevant exposure pathways can be chosen in a site-specific assessment to construct the scenario most suitable for the individual(s) of interest.

2.5.3.3. INPUT VARIABLES FOR EXPOSURE VIA SURFACE WATER -- A surface water body can be a lake, river, pond or reservoir, and can be used for fishing, swimming and boating, or as a source of drinking water.

Exposure to the contaminant if the surface water body is used as a drinking water source is affected by the concentration of the contaminant in the water, daily amount of water ingested and the percentage of time the individual spends in the area serviced by that water supply system. It is assumed in this methodology that treatment processes for drinking water do not alter the deposited contaminants. Chapter 10 further discusses determination of exposure from water ingestion.

Factors that affect human exposure by ingestion of fish from a contaminated surface water body include the sediment and water contaminant concentration, the types of fish/shellfish consumed and the ingestion rates for the various fish and shellfish groups, the bioconcentration factor for these groups, and the percent of the dietary fish that is caught in the surface water body near the source. The types of fish consumed will affect exposure because different types of fish and shellfish take up contaminants at varying rates. For example, fatty fish will tend to accumulate organic contaminants more readily than lean fish.

The amount of fish consumed also affects exposure as people who eat large amounts of fish will tend to have higher exposures. Fish consumption rates vary greatly depending on geographic region and social/cultural factors. Because 100% of an individual's dietary fish may not originate from the surface water body near the combustion facility, the percentage of locally caught fish is also a variable for exposure. Specific values for fish consumption rates are discussed in Chapter 11.

If the surface water body is used for recreational purposes such as swimming and boating, dermal absorption of contaminated water becomes another possible route for human exposure. The surface area of exposed skin, the chemical concentration in the water, the permeability of the skin to the chemical, and the length of time the individual is in contact with the water are the variables that contribute to dermal exposure. Incidental swallowing of water when using the water body for recreation also contributes to exposure. Dermal exposure via water is discussed in more detail in Chapter 12.

The input variables for various pathways of exposure via surface water can be changed to determine a range of exposures. An average exposure scenario might be represented by an individual that fishes, swims and obtains drinking water from the same water source (e.g., a large river or lake), while a worst-case possibility could be a person whose drinking water source is a cistern holding collected precipitation (see Section 2.5.3.4) and who fishes and swims in a small farm pond.

2.5.3.4. INPUT VARIABLES FOR PRECIPITATION FOR DRINKING WATER -- In several regions, rain and/or snow fall are collected and stored as a source of drinking water. Individuals in these areas may be exposed to pollutants deposited on the surface area from which water is collected. The input variables for this exposure pathway include: total annual deposition (both wet and dry deposition), annual precipitation, and the daily amount of water consumed. Section 9.3 further discusses this exposure pathway.

2.5.3.5. INPUT VARIABLES FOR GROUNDWATER INFILTRATION MODEL -- As a consequence of emitted pollutants infiltrating into the groundwater, the groundwater may be a source of exposure. The methodology developed to calculate risks from the groundwater pathway was originally intended to evaluate impacts from the landfilling of municipal sludge. Application of the methodology to deposited combustor emissions, showed it not to be a significant exposure pathway. This analysis is presented in Section 9.4.

2.6. DEFINING EXAMPLE EXPOSURE SCENARIOS

In this document, three example scenarios will be constructed and used to illustrate calculation of exposures. Risk assessors can vary the set of conditions to create exposure scenarios for a site-specific assessment. A hypothetical municipal waste combustor located in western Florida will be used as an example combustor facility. Scenario "A" is constructed to represent the most likely occurrence for an exposed population anywhere within a 50 km radius of the facility; typical or average values of input variables are chosen. In contrast, Scenario "C" illustrates the hypothetical case wherein all variables are maximized so that the highest potential exposure to an individual is determined. While most persons in the vicinity of a combustion facility are not likely to have such an exposure, this possibility may exist and should be explored. Scenario "B" strikes an approximate midpoint between Scenarios A and C; exposures will be higher than those in Scenario A but not as but infrequent as those in Scenario C.

For illustrative purposes in this document, certain assumptions will be made about the various routes of exposure. For example, for the food ingestion pathways, all milk will be assumed to be produced within the 50 km radius of the combustor facility. Depending on the exposure scenario, some milk may also be produced at distances closer to the facility. For all three hypothetical exposure scenarios, meat and food crops will be assumed to come from outside the vicinity of the combustor, unless produced by the individual. General descriptions of the three scenarios are provided below and in Table 2-3. Further details and example calculations for the scenarios are presented in each exposure pathway chapter.

2.6.1. Exposure Scenario A. Scenario A represents the most likely occurrence for an exposed population anywhere within a 50 km radius of the facility. This scenario is represented by an adult living in a metropolitan area (i.e., central city) where a MWC has been operating continuously for 30 years. Ground level concentrations of the emitted pollutants are relatively low, being represented as the average concentration for a 50 km (radial distance) ring around the source. The individual resides in this area for 16 years, but works (8 hours/day) away from the area receiving fallout.

In this scenario, a majority of the food is produced outside the vicinity with the exception of milk, which comes from a nearby dairy also in the area of average deposition. The individual grows some of his food in a small home garden, so that some of the diet is contaminated by the deposited emissions. The proportion

TABLE 2-3

Description of Exposure Scenarios

	Scenario A	Scenario B	Scenario C
Length Time of Combustor Emissions (years)	30	60	100
Location of Individual (km)	50 (central city)	5 (suburban)	0.2 (non-metropolitan)
Age	Adult	Child then adult	Child then adult
BW* (kg)	70	17 then 70	17 then 70
Years at Location	16	30	70
Food Production (meat and vegetables)	Percent home produced for central city dweller; remainder from outside 50 km radius of facility	Percent home produced for suburban dweller; remainder from outside 50 km radius of facility	Percent home produced for non-metropolitan dweller; remainder from outside 50 km radius of facility
Milk Production	100% within 50 km radius of facility	Percent home produced for suburban dweller (5 km); remainder within 50 km of facility	Percent home produced for non-metropolitan dweller (0.2 km); remainder within 50 km of facility
Food Consumption Rate (percentile)	50	70-75	90-95
Dermal Exposure from Soil (percentile)	Average	Upper end of average range	High-end of range

TABLE 2-3 (cont.)

	Scenario A	Scenario B	Scenario C
Soil Ingestion	≈ Average	≈ Upper-end of average range	Pica child; adult high-end of range
Water Source	Lake within 50 km	Lake within 50 km	Residential cistern
Fish Ingestion	≈ Average consumption for U.S. population	≈ 95th percentile for U.S. population	≈ 75th percentile for recreational fisherman
Water Ingestion Rate	≈ Average U.S. population	≈ 70-75 percentile for U.S. population	≈ 90-95 percentile for U.S. population
Dermal Exposure from Water	Not quantified	Not quantified	Not quantified

*BW of children may vary and is further discussed in Section 2.4.3.

of the meat and vegetables home grown will be estimated using the rates for center city dwellers as determined in the 1977-1978 USDA Food Consumption Survey, with the remainder of these food groups coming from outside the 50 km deposition radius, and therefore not contaminated by emitted pollutants. Specific inputs for exposure via food ingestion are discussed in Chapter 5. In general, the 50th percentile of consumption for all food groups will be used for this scenario.

The dermal exposure and soil ingestion pathways are relevant in this scenario since this person gardens. Average rates of soil ingestion and dermal exposure are used in this scenario. Specific inputs are detailed in the respective chapters.

A lake located in the vicinity provides drinking water for the metropolitan area and is used for fishing and recreation. This surface water body receives runoff and pollutants directly deposited from the MWC. While this person consumes average amounts of fish (average for general U.S. population), none are caught from the contaminated lake. Water ingestion rates for this individual are equal to the U.S. population average. Specific inputs for determining exposure via drinking and recreational usage of this water body are described in Chapters 10, 11 and 12.

2.6.2. Exposure Scenario B. Scenario B strikes a midpoint between Scenarios A and C and is represented by a child living in the suburbs, 5 km from the combustion facility, where the areal average ground-level concentrations of emissions (average for the 5 km ring surrounding the source) have been deposited for the 60 years of operation of the MWC. The child grows up and remains in this area for a total of 30 years. During childhood, the individual stays at home during the day, but as an adult works 8 hours/day away from the residence.

Some of this individual's food, i.e., milk, meats and vegetables, will be assumed to be home-grown and will therefore originate within the 5 km radius of the facility where the individual resides. Rates for the suburban area, as delineated in the 1977-1978 USDA Food Consumption Survey, will be used to estimate the proportion home-grown. The remainder of the milk will originate from within the 50 km radius of deposition, but the remaining proportion of meat and vegetable in the diet will be imported into the area. The upper end of the range of mean values for food consumption rates (approximately 70-75th percentile) of all food groups will be assumed.

Dermal exposure and soil ingestion pathways are also means of exposure since this person gardens. The upper end of the average rates, if available, of soil ingestion and dermal exposure will be used in this scenario for both the childhood and adulthood portions of this scenario. Specific inputs are detailed in the respective chapters.

A surface water body (lake) located in the area of moderate fallout is used as a drinking water source as well as for recreational swimming and fishing. All of the drinking water and some of the fish consumed by this individual are contaminated. It is assumed that the same drinking water source serves the individual's residence and place of work. The water ingestion rate for this individual is the 70-75th percentile for the U.S. population. The fish ingestion rate for this individual is the upper-end of the range for the U.S. population.

Specific input values used for this exposure scenario are described in the appropriate pathways.

2.6.3. Exposure Scenario C. Scenario C illustrates a case in which all variables are maximized so that the highest potential exposure can be determined. In exposure Scenario C, the individual is a child living in close proximity to the facility (0.2 km away) in a non-metropolitan area after the combustion source has been operating for 100 years. The child grows up and spends his lifetime (70 years) in this area. It is assumed the individual's daily activity is confined to the area of maximal deposition of pollutants.

In this scenario, a proportion of the dairy products will be come from the area of maximal deposition within the 0.2 km radius of the facility. Most of the meat and vegetables consumed by this individual will be home grown, with the remainder of the diet being imported into the area. The proportion of the diet grown by this individual will be estimated using rates for non-metropolitan area as stated in USDA Food Consumption Survey (1977-1978). Food consumption rates for this individual are assumed to be in the upper 90-95th percentile for all food groups.

As a youngster, this person has pica, and therefore, has high ingestion and dermal contact with soil. As an adult, the individual also has the high range of soil ingestion and dermal contact.

The individual obtains drinking water from a residential cistern which has collected precipitation. The water ingestion rate for this individual is the 90-95th percentile for the U.S. population. A pond, located near the residence, is the source of all fish consumed by this person, and is also used for swimming. It is assumed that this person consumes the mid-range values for fish for recreational fishermen. Chapters for each pathway further elaborate this scenario.

The remainder of this document presents a methodology for determining human exposure by various pathways, and then approaches for assessing if the exposure poses a risk to human health. Example calculations for cadmium and benzo(a)pyrene are presented for the hypothetical Scenario B.

3. AIR DISPERSION AND DEPOSITION OF EMITTED POLLUTANTS

3.1. BACKGROUND AND PURPOSE

Combustion of materials produces residual amounts of pollution that may be released to the environment. Estimation of potential human health risks associated with these releases requires knowledge of atmospheric pollutant concentrations and annual deposition rates in the areas around the combustion facility. Values for these quantities are most often estimated through the use of atmospheric dispersion models. To achieve the most realistic answers possible, these models should contain state-of-the-art algorithms to account for the effects of complex terrain and the process of pollutant deposition. In many instances, however, models that have very basic means of accounting for these two processes are used in risk assessments of municipal waste combustors. For example, the ISCST model provides an adequate treatment of flat terrain, but not for estimating the effects of deposition (U.S. EPA, 1986b). Other risk assessments for incinerators located in complex terrain (U.S. EPA, 1987b) have used the ISCLT and LONGZ models. In order to determine how appropriate the use of these simple models in situations where complex terrain effects and pollutant deposition are considered, it was necessary to compare results from these models with those generated by a complex terrain, deposition model. No such model is currently available through the U.S. EPA (1986c). Therefore, an appropriate complex terrain atmospheric dispersion model had to be selected that could be modified to realistically account for depositional effects. Two existing models, RTDM and COMPLEX I were altered to meet this requirement. This chapter evaluates the impact that the new models' treatment of complex terrain and depositional effects has on values used in risk assessments, compares their results with those from an existing model, and presents the areal-average deposition used in the three exposure scenarios A, B, and C in this document.

Deposition values for pollutants are the initial input for all models to estimate human exposure and risk. The choice of the appropriate model for air dispersion and deposition of emitted pollutants can, therefore, have a significant impact on the exposure scenario. The approaches shown here are illustrative. The risk assessor should evaluate the terrain, meteorologic conditions and other conditions for a site-specific assessment and determine the most appropriate model for the site.

3.2. MODEL SELECTION

Selection of appropriate complex terrain atmospheric dispersion models involved meeting three criteria: (1) account for the effects of complex terrain; (2) simulate yearly time periods (ideally five years or more); and (3) calculate both pollutant concentration and atmospheric deposition. A number of existing atmospheric models were investigated for possible use. These models generally fell into two categories.

The first group includes Gaussian puff and numerical simulation models. While many of these models contain algorithms for estimating pollutant deposition as well as concentration, they are usually very complex and sophisticated. The chief advantage of these models is their ability to calculate realistic wind fields, which in theory enable them to more effectively account for the effects of complex terrain. Further, many of these models are formulated such that these wind fields allow pollutant concentrations to be calculated at a specific time and location; thus, they are often used for simulating exposures during emergency situations. Generating wind fields, however, requires extensive computing time; consequently, these models are usually only viable for simulating time periods ranging from hours to a few days.

The second group contains less sophisticated models based on the Gaussian plume algorithm. Instead of calculating detailed wind fields, these models make only crude assumptions regarding complex terrain effects and are therefore better suited for determining average, time-independent concentrations at a desired location. Since these models are also less computer intensive than the first group of models, they may be used to simulate periods of a year or more. Like the more complicated models of the first group, many of these models also include provisions for estimating particle deposition.

The U.S. EPA Guideline on Air Quality Models and accompanying Supplement A (U.S. EPA, 1986c) contain recommendations for selecting models for use in complex terrain. No models capable of generating their own wind fields are recommended. Instead, three categories of screening models, initial, level two, and level three, are identified. The recommended level two and level three models in the Guideline and Supplement A were COMPLEX I and RTDM, respectively. COMPLEX I is a Gaussian plume model that incorporates a number of traditional algorithms for accounting for complex terrain effects. RTDM is a somewhat more advanced model that employs partial plume reflection algorithms, which prevent overestimates of pollutant concentrations near sloping terrain. Both of these models were selected for use in this analysis. Both of these models are Gaussian plume and are therefore well-suited for simulating time periods of one to five years. Algorithms used in COMPLEX I and RTDM to estimate air concentrations of pollutants are derived based on the assumption that no deposition occurs. However, for this analysis, it was also necessary to calculate pollutant deposition. Since concentration is dependent on deposition (i.e., plume depletion), accurate representations of both these quantities cannot be obtained unless the model used to calculate them contains concentration algorithms that consider depositional effects. Therefore, concentration algorithms explicitly formulated to account for the effects of deposition (U.S. EPA, 1982a) were substituted for the concentration algorithms originally found in COMPLEX I and RTDM to form two new models, COMPDEP and RTDMDEP. In addition, algorithms for estimating the effects of building wakes were obtained from a fourth model, ISCST, and used in the COMPDEP model. Details of each model are discussed in the following sections.

3.3. MODEL DESCRIPTIONS

3.3.1. General Model Development. Changes to existing models and new features incorporated in the COMPDEP and RTDMDEP models are given in these sections. For a more detailed description of existing models not included in this report, readers are referred to the user's guides for previously mentioned models: Turner (1986), U.S. EPA (1979), U.S. EPA (1980a), U.S. EPA (1982a), and ERT (1987).

3.3.2. COMPDEP Model. COMPDEP is a modification of COMPLEX I to account for both wet and dry deposition. The capability to estimate pollutant deposition was added to COMPLEX I by the inclusion of concentration algorithms from U.S. EPA (1982a). The algorithms from U.S. EPA (1982a) estimate pollutant deposition based on a single deposition velocity that is independent of particle size or atmospheric conditions. Therefore, to provide a more realistic value, these algorithms were altered to allow calculation of deposition velocity based on particle size and atmospheric conditions (CARB, 1986). Additional routines to enable the model to estimate pollutant concentration and deposition during periods of precipitation were also included. COMPDEP uses hourly meteorologic data as described in Section 3.4.2 to estimate atmospheric concentrations and pollutant deposition rates at desired receptor locations.

3.3.2.1. CONCENTRATION ALGORITHMS -- Hourly pollutant concentrations were calculated using algorithms taken from the MPTER-DS model. The equation used to calculate concentrations during time periods of no precipitation is given by:

$$Cd_i = \frac{FDRY \cdot F_i \cdot Q}{\sqrt{2\pi}\, U \cdot \sigma_y} \cdot [\exp -\tfrac{1}{2} \cdot (y/\sigma_y)^2] \cdot VDF \qquad \text{(Equation 3-1)}$$

where:

Cd_i = Pollutant concentration associated with particles of the ith size category during the portion of the during the portion of the hour in which no precipitation occurs (g/m^3)
FDRY = Correction factor for the fraction of the hour in which no precipitation occurs. Equal in magnitude to 1.0 - FWET
F_i = Fraction of pollutant attached to particles in the ith size category
Q = Source emission (g/s)
U = Mean wind speed at plume height and receptor location (m/s)
y = Horizontal crosswind distance from plume centerline to receptor (m)
σ_y = Gaussian plume dispersion parameter for the horizontal direction (m)
VDF = Vertical distribution function given by one of the following equations:

For stable conditions or unlimited mixing:

$$\begin{aligned} VDF = \frac{1}{\sqrt{2\pi}\sigma_z} \cdot [&\exp\{-2\hat{W} \cdot (\hat{z} - \hat{H}) \cdot \hat{x} - x_1^2\} \cdot \\ &[\exp\{-(\hat{z} - \hat{H})^2\} + \exp\{-(\hat{z} + \hat{H})^2\} \cdot \\ &\{1 - 4\sqrt{\pi} \cdot V_1 \cdot \hat{x} \cdot \exp(\xi^2) \cdot \mathrm{erfc}(\xi)\}]] \end{aligned} \qquad \text{(Equation 3-2)}$$

For unstable or neutral conditions with $\sigma_z < 1.6L$

$$\begin{aligned} VDF = \frac{1}{\sqrt{2\pi}\sigma_z} \cdot \sum_{N=-\infty}^{\infty} [&\exp\{-2\hat{W} \cdot (\hat{z} - \hat{H}_1) \cdot \hat{x} - x_1^2\} \cdot \\ &[\exp\{-(\hat{z} - \hat{H}_1)^2\} + \exp\{-(\hat{z} + \hat{H}_1)^2\} \cdot \\ &\{1 - 4\sqrt{\pi} \cdot V_1 \cdot \hat{x} \cdot \exp(\xi_1^2) \cdot \mathrm{erfc}(\xi_1)\}]] \end{aligned} \qquad \text{(Equation 3-3)}$$

For stable or neutral conditions with uniform vertical mixing $\sigma_z \geq 1.6L$ and $V_2 = 0.0$

$$VDF = \frac{1}{L} \cdot [(1 + 2x_1^2) \cdot \mathrm{erfc}(x_1) - (2x_1/\sqrt{\pi}) \cdot \exp(-x_1^2)]$$ (Equation 3-4)

For $V_2 \neq 0.0$

$$VDF = \frac{1}{L} \cdot [V_1/V_2) \cdot \exp(4\hat{V}_d \cdot V_2 \cdot \hat{x}^2) \cdot (2V_1 \cdot \hat{x}) - (\hat{W}/2V_2) \cdot \mathrm{erfc}(x_1)]$$ (Equation 3-5)

where:

$\hat{H} = HE/\sqrt{2}\sigma_z$ (Equation 3-6)

$H_1 = HE + 2N \cdot L$ (Equation 3-7)

$\hat{H}_1 = H_1/\sqrt{2}\sigma_z$ (Equation 3-8)

$\hat{x} = x/\sqrt{2}\sigma_z$ (Equation 3-9)

$x_1 = \hat{x} \cdot \hat{W}$ (Equation 3-10)

$\hat{z} = z/\sqrt{2}\sigma_z$ (Equation 3-11)

$\hat{W} = W/U$ (Equation 3-12)

$\hat{V}_d = V_d/U$ (Equation 3-13)

$V_1 = \hat{V}_d - \hat{W}/2$ (Equation 3-14)

$V_2 = \hat{V}_d - \hat{W}$ (Equation 3-15)

$\xi = \hat{z} + \hat{H} + 2V_1 \cdot \hat{x}$ (Equation 3-16)

$\xi_1 = \hat{z} + \hat{H}_1 + 2V_1 \cdot \hat{x}$ (Equation 3-17) and

HE = Effective height of source emission (m)
N = Eddy reflection number
L = Mixing depth of inversion lid (m)
x = Horizontal upwind distance of source from receptor (m)
z = Vertical coordinate of receptor (m)
W = Gravitational settling velocity (m/s)
V_d = Dry deposition velocity (m/s)

Effects of dry deposition and gravitational settling are accounted for in the vertical distribution function through the use of deposition and gravitational settling velocities. For the fraction of the hour during which there is no precipitation, concentrations are calculated using deposition and settling velocities described in Section 3.3.2.2.

Concentrations are also calculated for periods in which precipitation occurs. For the fraction of the hour during which precipitation takes place (identified according to methods described in Section 3.4.2.2), concentrations are calculated with deposition and gravitational velocities set equal to zero. This is based on the assumption that the effects of dry deposition and gravitational settling are negligible compared with precipitation scavenging. Effects of precipitation scavenging during these periods are approximated by the addition of an exponential loss term to the equation used to calculate pollutant concentrations:

$$Cw_i = \frac{FWET \cdot F \cdot Q}{\sqrt{2\pi}\, U \cdot \sigma_y} \cdot [\exp\{-\tfrac{1}{2}(y/\sigma_y)^2\}] \cdot VDF \cdot \exp(-\Lambda \cdot x/U) \qquad \text{(Equation 3-18)}$$

where:

Cw_i = Pollutant concentration in the portion of the hour in which precipitation does occur associated with particles in the ith particle size category (g/m^3)
Λ = Scavenging coefficient (1/s)

The model uses a different scavenging coefficient for each particle size and precipitation intensity.

The total hourly concentration of pollutant is the sum of the dry and wet pollutant concentrations (Cd_i and Cw_i, respectively) over all particle size categories.

3.3.2.2. POLLUTANT DEPOSITION -- Both dry and wet deposition are calculated by the model. During periods in which no precipitation occurs, dry deposition of pollutant associated with each particle size is calculated as the product of concentration (as determined by Equation 3-1) and a particle size dependent deposition velocity. This velocity is automatically calculated by the model from routines developed by the California Air Resources Board (CARB, 1986). These algorithms were selected based on results from an evaluation of four deposition velocity models (Travis and Yambert, 1989), which showed them to be superior in estimating deposition velocities of particulates to grasses and small crops, the type of surfaces in risk assessments for which deposition estimates are often desired. Total dry deposition for the period, $DEPd_i$ ($g/m^2/s$) is obtained by summing the depositions for each different particle size.

Using methods given in Slinn (1984), wet deposition is estimated by integrating the product of Equation 3-17 and the scavenging coefficient, Λ, over the range of $0.0 < z < \infty$. The resulting equation used to calculate wet deposition for particles in each size category is:

$$DEPw_i = \frac{\Lambda \cdot FWET \cdot F_i \cdot Q}{\sqrt{2\pi}\, U \cdot \sigma_y} \cdot \exp[-\tfrac{1}{2}(y/\sigma_y)^2] \cdot \exp(-\Lambda \cdot x/U) \qquad \text{(Equation 3-19)}$$

where:

$DEPw_i$ = Hourly wet deposition value for associated particles in the ith size category ($g/m^2/s$)

Total wet deposition for each hour is the sum of the values for each particle size.

3.3.2.3. TERRAIN REPRESENTATION -- COMPLEX I contains five different methods for estimating the effects of complex terrain. The first of these algorithms was retained for use in the COMPDEP model (U.S. EPA, 1986b). Effects of complex terrain are accounted for in the following manner. First, the effective plume height is equivalent to the maximum of the following calculated values:

$$HE = H - THT \cdot (1.0 - TER) \qquad \text{(Equation 3-20)}$$

$$HE = H \cdot TER \qquad \text{(Equation 3-21)}$$

$$HE = HMIN \qquad \text{(Equation 3-22)}$$

where:

H = Plume height (sum of stack height and plume rise) (m)
THT = The height of the terrain obstacle above stack base elevation (m)
TER = Terrain adjustment factor ranging in value from 0.0 to 1.0
HMIN = Minimum distance between plume and ground (m) (usually set to 10.0 m)

For receptors whose ground level elevation exceeds the plume elevation (the sum of stack base elevation and plume height), pollutant concentrations and depositions are multiplied by the following correction factor:

$$\mathrm{CORR} = (400.0 - \mathrm{DIFF})/400.0 \qquad 0.0 < \mathrm{DIFF} < 400.0\ \mathrm{m} \qquad \text{(Equation 3-23)}$$

$$= 0.0 \qquad \mathrm{DIFF} > 400.0\ \mathrm{m} \qquad \text{(Equation 3-24)}$$

where:

DIFF = The difference between receptor ground level elevation and plume elevation (m)

3.3.2.4. BUILDING WAKE EFFECTS -- COMPLEX I does not contain a methodology for estimating building wake effects. To provide the model with this capability, algorithms from the ISCST model were used.

3.3.3. RTDMDEP. RTDMDEP is a modification of RTDM to account for both wet and dry deposition. RTDM is recognized as a third-level screening model for use in complex terrain. Like COMPLEX I, it was modified to account for the effects of pollutant deposition by the inclusion of concentration algorithms described in U.S. EPA (1982a). The modifications performed are identical to those described in Sections 3.3.2.1 and 3.3.2.2 for the COMPDEP model. The new model, RTDMDEP, calculates pollutant concentrations and depositions during periods of precipitation and no precipitation using deposition velocities that are particle size dependent (CARB, 1986). With the exception of these modifications, RTDMDEP is very similar in form to the original RTDM model. The chief difference between it and COMPDEP is the way in which the model accounts for the effects of complex terrain. This difference is discussed briefly in the next section.

3.3.3.1. TERRAIN REPRESENTATION -- Like COMPDEP, RTDMDEP modifies the effective plume height to account for the effects of terrain; however, it refines this approach with the additional consideration of critical height defined as:

$$Hcrit = THT \cdot (1.0 - Fr) \qquad \text{(Equation 3-25)}$$

where:

Hcrit = Critical plume height defined such that only air above this level will pass over a terrain feature (m)
Fr = Froude number defined as:

$$Fr = u/(THT \cdot \sqrt{(g \cdot VPTG/Ta)}) \qquad \text{(Equation 3-26)}$$

where:

u = Stack top wind speed (m/s)
g = The acceleration of gravity (9.8 m/s^2)
VPTG = Vertical potential temperature gradient (K/m)
Ta = Ambient temperature (K)

The effective plume height is calculated according to:

$$HE = TER \cdot Hpc \qquad H \leq THT \qquad \text{(Equation 3-27)}$$

$$HE = Hpc - (1.0 - TER) \cdot Htc \qquad H > THT \qquad \text{(Equation 3-28)}$$

where:

Hpc = The height of the plume above the critical height (m)
Htc = The height of the local terrain above the critical height (m)

Additionally, to avoid artificial plume depression in cases where the plume is above the critical height, RTDMDEP assumes a level plume by setting the value for TER equal to zero. RTDMDEP also adjusts the mixing height in an analogous manner to that used in calculating effective plume height.

RTDMDEP contains an algorithm used to modify pollutant concentrations to account for the effects of partial plume reflection from sloping terrain. Conventional Gaussian plume models model the ground surface through the use of "image source" routines. These routines can result in unrealistically high estimated pollutant concentrations for cases in which a plume approaches rapidly sloping terrain. The partial plume reflection algorithm used in RTDMDEP prevents these inaccuracies by multiplying pollutant concentrations by a reflection factor, R, based on terrain slope and plume growth. This factor is the ratio of the minimum value of the maximum cross wind integrated concentration (MCWI) to the cross wind integrated concentration evaluated at the point of impact of closest plume approach to terrain.

3.3.4. ISCST. The Industrial Source Complex Short Term Model (ISCST) is a preferred model for use in flat or moderately rolling terrain when modeling sources such as stacks from MWCs or other industrial source complexes (U.S. EPA, 1986c). It is a steady state Gaussian plume model that contains limited provisions for estimating the effects of terrain on pollutant transport. ISCST is sometimes used to estimate pollutant transport in complex terrain. However, concentrations estimated for points above stack top elevation are subject to extreme uncertainty as the model substitutes stack top elevation for the point's actual elevation. Pollutant deposition is calculated through the use of surface reflection coefficients and gravitational settling velocities.

3.4. ANALYSIS PROCEDURES

The modeling analysis conducted was designed with two primary goals. The first of these was a quantification of how atmospheric pollutant concentrations predicted by the two models detailed in this paper differ from those calculated by existing flat terrain models for sources located in complex terrain. Second, an evaluation of the deposition velocity based algorithms used by COMPDEP and RTDMDEP was desired. In order to accomplish this, model runs were made for a hypothetical, complex terrain incinerator using COMPDEP, RTDMDEP, and the existing flat terrain model, ISCST. ISCST was chosen for the analysis since it is probably the most widely used and accepted flat terrain model for estimating pollutant concentrations due to stack emissions.

Whereas terrain elevation is the chief aspect of complex terrain that affects predictions made by COMPDEP, concentration predictions made by RTDMDEP are affected by terrain slope and hill height as well. Because of this difference, two receptor patterns were considered. Concentrations predicted by COMPDEP and ISCST were calculated for a variety of terrain elevations, with no consideration given to terrain slope. RTDMDEP model runs analyzed selected COMPDEP and ISCST receptor locations but incorporated terrain slope and hill height into model calculations in addition to terrain elevation.

Each model was run in two modes. The first of these determined atmospheric pollutant concentrations assuming no deposition occurred, while the second calculated both atmospheric pollutant concentrations and pollutant deposition rate. Results from these analyses are presented in Sections 3.5.0 - 3.5.3.2. Model input is described in the following sections.

3.4.1. Source Characteristics. The source parameters for a hypothetical incinerator in the Northeastern United States used for comparison of the models study, given in Table 3-1, are taken from an existing mass burn incinerator. Unit emission rates of pollutant were used for all model runs. For model runs that considered pollutant deposition, emitted pollutants were assumed to fall into one of three particle size categories: (1) less than 2 μm, (2) 2 to 10 μm, and (3) greater than 10 μm. Representative sizes and fraction of emissions in each of these categories were based on data for organic pollutants found in U.S. EPA (1986b) and are, respectively, 1.0 μm and 0.875 for category 1, 6.78 μm and 0.095 for category 2, and 20.0 μm and 0.030 for category 3.

3.4.2. Meteorologic Data. In order to realistically consider possibly occurring meteorologic conditions, model runs were made using one year of actual meteorologic data for Albany, New York. The Source Receptor Analysis Branch of the U.S. EPA at Research

TABLE 3-1

Source Characteristics for MWC Analysis

Parameter	Value
Stack diameter (m)	1.04
Stack exit velocity (m/s)	15.24
Stack gas temperature (K)	327.60
Building Dimensions (m)	
Height	11.00
Width	48.80
Length	73.20

Triangle Park, NC provided this data in standard, preprocessed format containing hourly values for mixing height, stability class, temperature, wind direction, and wind speed.

3.4.3. Receptor Locations. Receptor locations used by ISCST and COMPDEP model runs were chosen to reflect how model predictions are influenced by terrain both near to and far away from the source. Whereas terrain elevation is the chief aspect of complex terrain, which affects predictions made by COMPDEP and ISCST, predictions made by RTDMDEP are also influenced by hill height. Because of this difference, two receptor patterns were considered. Predictions made by COMPDEP and ISCST were calculated for a variety of terrain elevations, with no consideration given to terrain slope. Receptor distances for these model runs varied from 200 to 50,000 meters, while elevations ranging from that of the stack base to 300 meters above the top of the stack were considered. The most frequent wind direction for all stability classes during 1970 was due north (U.S. EPA, 1987b). Thus, to ensure exposure to the widest possible range of meteorologic conditions, all receptors were located along this direction.

Selected receptor locations analyzed by COMPDEP and ISCST were also analyzed by RTDMDEP. In order to investigate the influence of terrain slope on model predictions, three sets of RTDMDEP runs were made assuming terrain slopes of 30.0°, 45.0°, and 60.0° respectively.

The coordinate systems used to express receptor locations are defined as follows. Coordinate systems for both locations are rectangular with the "+x" direction corresponding to east, and the "+y" direction corresponding to north. Origins for all systems correspond to the stack location. A complete listing of receptor locations used by ISCST and RTDMDEP is given in Table 3-2. Receptor locations used by RTDMDEP for each of the three terrain slopes considered are found in Table 3-3.

3.4.4. Model Run Descriptions. For each model, a concentration-only run, and a deposition run were made. For all deposition runs, only dry deposition was calculated. In its original form, ISCST does not determine wet deposition. Consequently, a comparison of wet depositions between flat and complex terrain models could not be attained without its modification. Hence, for purposes of this analysis, wet deposition was not determined.

The following were assumed by all model runs: exponents for power-law wind increase with height of 0.07, 0.07, 0.10, 0.15, 0.35, 0.55, anemometer height of 10.0 meters, surface roughness of 0.10 meters, no pollutant decay. Model options specific to each of the three models used in the analysis are presented in Tables 3-4 to 3-6.

3.4.5. Reflection Coefficients Used by ISCST. The algorithms used by COMPDEP and RTDMDEP to estimate pollutant deposition require an estimate of deposition velocity for each particle size analyzed. This is done automatically via the inclusion of the CARB (1986) algorithms. Unlike these models, ISCST requires that a reflection coefficient and gravitational settling velocity be input manually

TABLE 3-2

Receptor Locations Used by COMPDEP and ISCST

Receptor Number	X Coordinate (m)	Y Coordinate (m)	Elevation (m)
1	0.0	200.0	169.0
2	0.0	400.0	169.0
3	0.0	500.0	169.0
4	0.0	600.0	169.0
5	0.0	800.0	169.0
6	0.0	1000.0	169.0
7	0.0	1250.0	169.0
8	0.0	1500.0	169.0
9	0.0	1750.0	169.0
10	0.0	2000.0	169.0
11	0.0	2500.0	169.0
12	0.0	3000.0	169.0
13	0.0	4000.0	169.0
14	0.0	5000.0	169.0
15	0.0	10000.0	169.0
16	0.0	20000.0	169.0
17	0.0	30000.0	169.0
18	0.0	40000.0	169.0
19	0.0	50000.0	169.0
20	0.0	200.0	194.0
21	0.0	400.0	194.0
22	0.0	500.0	194.0
23	0.0	600.0	194.0
24	0.0	800.0	194.0
25	0.0	1000.0	194.0
26	0.0	1250.0	194.0
27	0.0	1500.0	194.0
28	0.0	1750.0	194.0
29	0.0	2000.0	194.0
30	0.0	2500.0	194.0
31	0.0	3000.0	194.0
32	0.0	4000.0	194.0
33	0.0	5000.0	194.0
34	0.0	10000.0	194.0
35	0.0	20000.0	194.0
36	0.0	30000.0	194.0
37	0.0	40000.0	194.0
38	0.0	50000.0	194.0
39	0.0	200.0	220.0
40	0.0	400.0	220.0
41	0.0	500.0	220.0
42	0.0	600.0	220.0
43	0.0	800.0	220.0
44	0.0	1000.0	220.0
45	0.0	1250.0	220.0
46	0.0	1500.0	220.0
47	0.0	1750.0	220.0

TABLE 3-2 (cont.)

Receptor Number	X Coordinate (m)	Y Coordinate (m)	Elevation (m)
48	0.0	2000.0	220.0
49	0.0	2500.0	220.0
50	0.0	3000.0	220.0
51	0.0	4000.0	220.0
52	0.0	5000.0	220.0
53	0.0	10000.0	220.0
54	0.0	20000.0	220.0
55	0.0	30000.0	220.0
56	0.0	40000.0	220.0
57	0.0	50000.0	220.0
58	0.0	200.0	270.0
59	0.0	400.0	270.0
60	0.0	500.0	270.0
61	0.0	600.0	270.0
62	0.0	800.0	270.0
63	0.0	1000.0	270.0
64	0.0	1250.0	270.0
65	0.0	1500.0	270.0
66	0.0	1750.0	270.0
67	0.0	2000.0	270.0
68	0.0	2500.0	270.0
69	0.0	3000.0	270.0
70	0.0	4000.0	270.0
71	0.0	5000.0	270.0
72	0.0	10000.0	270.0
73	0.0	20000.0	270.0
74	0.0	30000.0	270.0
75	0.0	40000.0	270.0
76	0.0	50000.0	270.0
77	0.0	200.0	470.0
78	0.0	400.0	470.0
79	0.0	500.0	470.0
80	0.0	600.0	470.0
81	0.0	800.0	470.0
82	0.0	1000.0	470.0
83	0.0	1250.0	470.0
84	0.0	1500.0	470.0
85	0.0	1750.0	470.0
86	0.0	2000.0	470.0
87	0.0	2500.0	470.0
88	0.0	3000.0	470.0
89	0.0	4000.0	470.0
90	0.0	5000.0	470.0
91	0.0	10000.0	470.0
92	0.0	20000.0	470.0
93	0.0	30000.0	470.0
94	0.0	40000.0	470.0
95	0.0	50000.0	470.0

TABLE 3-3

Receptor Locations Used by RTDMDEP for Terrain Slope of 30, 45, and 60°

Receptor Number	X Coordinate (m)	Y Coordinate (m)	Elevation (m)
1	0.0	500.0	194.0
2	0.0	500.0	220.0
3	0.0	500.0	270.0
4	0.0	500.0	470.0
5	0.0	1000.0	194.0
6	0.0	1000.0	220.0
7	0.0	1000.0	270.0
8	0.0	1000.0	470.0
9	0.0	5000.0	194.0
10	0.0	5000.0	220.0
11	0.0	5000.0	270.0
12	0.0	5000.0	470.0

TABLE 3-4

COMPDEP Model Options

Building Wake Effects
Terrain Effects
Gradual Plume Rise
Buoyancy Induced Dispersion
Building Wake Effects
No Calm Processing
No Stack Downwash

TABLE 3-5

RTDMDEP Model Options

Terrain Effects

Gradual Plume Rise

Buoyancy Induced Dispersion

No Stack Downwash

Pasquill-Gifford Dispersion Coefficients

Partial Plume Penetration (VPTG 0.0060 Deg K/m)

Unlimited Mixing for Stable Conditions

Plume Path Coefficient for Stability Classes 1-6: 0.50

Default Vertical Potential Temperature Gradients for Stability Classes 5 & 6

Stability Class-Dependent σ_y and σ_z

No Wind Direction Shear

Partial Plume Reflection

Off Centerline Horizontal Distribution Function for All Stabilities

Constant Emission Rate

TABLE 3-6

ISCST Model Options

Terrain Effects
Rural Option
Gradual Plume Rise
Buoyancy Induced Dispersion
Building Wake Effects
No Stack Downwash
No Calm Processing
Default Vertical Potential Temperature Gradients

for each particle size. Values of these quantities used for all model runs were taken from U.S. EPA (1986b) and are listed in Table 3-7.

3.5. COMPARISON OF MODEL RESULTS

3.5.1. Calculated Air Concentrations Neglecting Deposition Effects.

3.5.1.1. COMPDEP AND ISCST RESULTS -- An examination of Table 3-8 results shows that for receptors 1 - 19, which correspond to flat terrain, COMPDEP and ISCST predict essentially equivalent atmospheric pollutant concentrations. This agrees with anticipated results since the concentration algorithms used by both models are similar for conditions in which the effects of terrain and pollutant deposition are neglected.

Once terrain elevations are introduced, differences in model predictions increase. For receptors with elevations of half the stack height, receptors 20 - 38, ISCST predictions exceed those of COMPDEP by a factor of 1.6 on an average basis and individually by as much as a factor of 5.2 (at receptor 1) at distances less than or equal to 10 km from the source. At distances greater than 10 km, both models again predict nearly identical concentrations.

A similar pattern is seen for receptors that are equal to the stack height in elevation (receptors 39 - 57). ISCST predictions for receptors within 10 km of the stack exceed those of COMPDEP by an average factor of 1.7. The maximum deviation between the two

TABLE 3-7

Reflection Coefficients and Gravitational Settling Velocities Used in ISCST Model Runs*

Particle Diameter (μm)	Reflection Coefficient	Settling Velocity (cm/s)
D ≤ 2.0	0.961	5.55E-03
2.0 ≤ D ≤ 10.0	0.861	2.55E-01
D ≥ 10.0	0.714	2.22E+00

*Source: U.S. EPA, 1986b

TABLE 3-8

Average Atmospheric Concentrations ($\mu g/m^3$), Neglecting Deposition Effects, Predicted by COMPDEP and ISCST

Receptor Number	COMPDEP Concentration	ISCST Concentration
1	0.848E-02	0.849E-02
2	0.112E+00	0.112E+00
3	0.165E+00	0.165E+00
4	0.212E+00	0.212E+00
5	0.295E+00	0.296E+00
6	0.350E+00	0.350E+00
7	0.355E+00	0.355E+00
8	0.342E+00	0.342E+00
9	0.322E+00	0.322E+00
10	0.300E+00	0.300E+00
11	0.258E+00	0.258E+00
12	0.223E+00	0.223E+00
13	0.171E+00	0.171E+00
14	0.137E+00	0.137E+00
15	0.640E-01	0.640E-01
16	0.284E-01	0.284E-01
17	0.174E-01	0.174E-01
18	0.124E-01	0.124E-01
19	0.949E-02	0.949E-02
20	0.404E-01	0.212E+00
21	0.271E+00	0.829E+00
22	0.381E+00	0.101E+01
23	0.474E+00	0.109E+01
24	0.589E+00	0.107E+01
25	0.622E+00	0.965E+00
26	0.581E+00	0.814E+00
27	0.532E+00	0.694E+00
28	0.484E+00	0.602E+00
29	0.441E+00	0.528E+00
30	0.367E+00	0.419E+00
31	0.310E+00	0.343E+00
32	0.231E+00	0.248E+00
33	0.181E+00	0.191E+00
34	0.795E-01	0.809E-01
35	0.332E-01	0.334E-01
36	0.199E-01	0.199E-01
37	0.139E-01	0.139E-01
38	0.106E-01	0.106E-01
39	0.117E+01	0.128E+02
40	0.132E+01	0.650E+01
41	0.148E+01	0.500E+01
42	0.156E+01	0.401E+01
43	0.153E+01	0.282E+01
44	0.139E+01	0.212E+01
45	0.118E+01	0.161E+01
46	0.101E+01	0.129E+01
47	0.870E+00	0.106E+01

TABLE 3-8 (cont.)

Receptor Number	COMPDEP Concentration	ISCST Concentration
48	0.757E+00	0.890E+00
49	0.589E+00	0.663E+00
50	0.473E+00	0.518E+00
51	0.330E+00	0.350E+00
52	0.247E+00	0.259E+00
53	0.966E-01	0.981E-01
54	0.377E-01	0.379E-01
55	0.219E-01	0.220E-01
56	0.152E-01	0.152E-01
57	0.114E-01	0.114E-01
58	0.851E+01	0.128E+02
59	0.569E+01	0.650E+01
60	0.489E+01	0.500E+01
61	0.422E+01	0.401E+01
62	0.320E+01	0.282E+01
63	0.248E+01	0.212E+01
64	0.189E+01	0.161E+01
65	0.149E+01	0.129E+01
66	0.121E+01	0.106E+01
67	0.100E+01	0.890E+00
68	0.733E+00	0.663E+00
69	0.564E+00	0.518E+00
70	0.374E+00	0.350E+00
71	0.271E+00	0.258E+00
72	0.992E-01	0.981E-01
73	0.374E-01	0.379E-01
74	0.216E-01	0.220E-01
75	0.148E-01	0.152E-01
76	0.111E-01	0.114E-01
77	0.403E+01	0.128E+02
78	0.270E+01	0.650E+01
79	0.231E+01	0.500E+01
80	0.199E+01	0.401E+01
81	0.151E+01	0.282E+01
82	0.117E+01	0.212E+01
83	0.896E+00	0.161E+01
84	0.702E+00	0.129E+01
85	0.570E+00	0.106E+01
86	0.473E+00	0.890E+00
87	0.346E+00	0.663E+00
88	0.266E+00	0.518E+00
89	0.176E+00	0.350E+00
90	0.128E+00	0.258E+00
91	0.468E-01	0.981E-01
92	0.177E-01	0.379E-01
93	0.102E-01	0.220E-01
94	0.702E-02	0.152E-01
95	0.527E-02	0.114E-01

models' predictions occurs 200 meters from the source where the ISCST concentration is a factor of 10.9 times that of COMPDEP. As with the previous two groups of receptors, for distant points (those in excess of 10 km from the source) concentration predictions are essentially identical. In summary, for the elevations considered so far, it is apparent that while average concentrations predicted by ISCST and COMPDEP agree rather well even in complex terrain, individual differences between ISCST and COMPDEP differ by as much as a factor of ten.

An examination of the remaining two groups of receptors considered, those that are 100 and 300 meters above stack height, yields results that differ somewhat from those obtained for the first three groups investigated. While concentrations predicted for receptors 100 meters above stack height and less than 600 meters from the source are 1.0 to 1.5 times higher than COMPDEP's, ISCST predictions are less than COMPDEP's, by an average factor of 0.9, between 600 and 20,000 meters from the stack. This discrepancy may be explained by the fact that, for terrain above the stack, ISCST substitutes the stack height for the actual terrain elevation in concentration calculations. Consequently, this approach yields almost random values, which should be considered highly uncertain. Concentrations predicted by COMPDEP are greater at this elevation than at stack height. This indicates that the terrain correction factor described in Section 3.3.2.3 has little effect at this elevation as the plume height (the sum of stack height and plume rise) is still above terrain elevation. ISCST concentrations beyond 20 km are again equal to those of COMPDEP.

In contrast to results for receptors 100 meters above stack elevation, concentrations predicted by ISCST at elevations 300 meters above stack top are greater on average than those of COMPDEP by a factor of 2.1 for all locations. The maximum deviation occurs at receptor 77 where ISCST overpredicts COMPDEP by a factor of 3.2. While concentrations predicted by ISCST are identical to those for the previous two groups, those calculated by COMPDEP are significantly less than at the previous elevation. Thus, for this elevation, COMPDEP's terrain correction factor has a considerable influence on model predictions.

3.5.1.2. RTDMDEP Results -- An examination of concentrations predicted by the third model used in the analysis, RTDMDEP, provides additional interesting results (See Table 3.9). As stated in Section 3.3.1, RTDMDEP employs a partial plume penetration algorithm that, for elevated terrain, tends to prevent erroneous magnification of pollutant concentrations. It is thus expected that concentrations predicted by RTDMDEP should be less than those of COMPDEP in areas of elevated terrain. For the most part, this is indeed the case. Excluding receptor 2, where processes that occur close to the source such as building wake effects, tend to dominate pollutant concentrations, values predicted by COMPDEP for receptors equal to and above stack height exceed those of RTDMDEP for all terrain slope values. For these points, COMPDEP

TABLE 3-9

Average Atmospheric Concentrations ($\mu g/m^3$), Neglecting Deposition Effects, Predicted by RTDMDEP, COMPDEP and ISCST

Receptor Number	RTDMDEP 30° Slope	RTDMDEP 45° Slope	RTDMDEP 60° Slope	COMPDEP	ISCST
1	0.395E+00	0.395E+00	0.395E+00	0.381E+00	0.101E+01
2	0.197E+01	0.197E+01	0.220E+01	0.148E+01	0.500E+01
3	0.242E+01	0.242E+01	0.242E+01	0.489E+01	0.500E+01
4	0.179E+01	0.179E+01	0.174E+01	0.231E+01	0.500E+01
5	0.662E+00	0.662E+00	0.662E+00	0.622E+00	0.965E+00
6	0.132E+01	0.132E+01	0.133E+01	0.139E+01	0.212E+01
7	0.143E+01	0.143E+01	0.143E+01	0.248E+01	0.212E+01
8	0.115E+01	0.104E+01	0.990E+00	0.117E+01	0.212E+01
9	0.176E+00	0.176E+00	0.176E+00	0.181E+00	0.191E+00
10	0.175E+00	0.178E+00	0.182E+00	0.247E+00	0.259E+00
11	0.173E+00	0.173E+00	0.174E+00	0.271E+00	0.258E+00
12	0.107E+00	0.108E+00	0.109E+00	0.128E+00	0.258E+00

predictions exceed those of RTDMDEP by a factor of 1.1 at receptor 6 to a value of 2.0 at receptor 3. These results are consistent with previous evaluations of COMPLEX I and RTDM (Paine and Egan, 1986) which show that both models have a tendency to overestimate pollutant concentrations in complex terrain, and that the atmospheric concentrations predicted by RTDM are less than those determined by COMPLEX I.

For receptors 1 and 5, which are lower than stack height and near the source, the COMPDEP concentration are less than that of RTDMDEP. This is most likely due to the fact that the partial plume reflection algorithm of RTDMDEP does not affect terrain below stack elevation.

Of further interest is the fact that for all receptors above stack height, differences in values predicted by the two models decrease with increasing receptor elevation. This, and the fact that the algorithms RTDMDEP uses to compute terrain effects are much more sophisticated than those in COMPDEP, suggests that the simple terrain correction factor (Section 3.3.1) used by COMPDEP works reasonably well for the range of elevations considered.

Variation of terrain slope seems to have a minimal effect on concentrations predicted by RTDMDEP for the conditions modeled. The greatest differences are seen at locations of large downwind distances and high elevations (Receptors 8, 10, 11, and 12,). The highest change occurs at Receptor 8 where a 13.9% decrease in concentration occurs as terrain slope varies from 30° to 60°. It is possible that additional variation in hill height used in model calculations could magnify the differences seen.

For all locations, concentrations predicted by ISCST are greater than those of RTDMDEP. ISCST predicts concentrations that range from 1.1 to 2.8 times those of RTDMDEP.

3.5.2. Calculated Air Concentrations Including Deposition Effects. ISCST is designed to provide calculated values for either atmospheric concentrations or pollutant deposition rate. However, both COMPDEP and RTDMDEP determine both quantities when the option to calculate pollutant deposition is selected. Tables 3-10 and 3-11 list atmospheric concentrations calculated by these two models when they are run in deposition mode. It can be seen that these concentrations are almost the same as those obtained when deposition effects are not considered. Only at distances greater than 5 km do the COMPDEP results differ significantly than those in Table 3-8, while RTDMDEP results are nearly identical for all locations. This is understandable when one considers that, over a period of time, deposition results in a depleted plume. Plume depletion is reflected by an initial decrease in pollutant concentration at the top of the plume that gradually extends downward as plume travel time increases. Plume travel time to receptors near the source pollutant is relatively short. Thus, while air concentrations near the top of the plume may be somewhat reduced, concentrations at ground level receptors, like those used in the analysis, are similar to when no deposition effects are considered. As seen by the results of this analysis, when plume

TABLE 3-10

Average Atmospheric Concentrations ($\mu g/m^3$) Including Dry Deposition Effects Predicted by COMPDEP

Receptor Number	COMPDEP Concentration
1	0.851E-02
2	0.112E+00
3	0.165E+00
4	0.212E+00
5	0.295E+00
6	0.349E+00
7	0.353E+00
8	0.340E+00
9	0.320E+00
10	0.298E+00
11	0.255E+00
12	0.220E+00
13	0.169E+00
14	0.134E+00
15	0.615E-01
16	0.262E-01
17	0.156E-01
18	0.108E-01
19	0.882E-02
20	0.405E-01
21	0.271E+00
22	0.381E+00
23	0.474E+00
24	0.589E+00
25	0.620E+00
26	0.579E+00
27	0.529E+00
28	0.481E+00
29	0.438E+00
30	0.363E+00
31	0.306E+00
32	0.227E+00
33	0.177E+00
34	0.751E-01
35	0.300E-01
36	0.174E-01
37	0.118E-01
38	0.874E-02
39	0.117E+01
40	0.132E+01
41	0.148E+01
42	0.156E+01
43	0.153E+01
44	0.139E+01
45	0.117E+01
46	0.100E+01
47	0.859E+00

TABLE 3-10 (cont.)

Receptor Number	COMPDEP Concentration
48	0.745E+00
49	0.576E+00
52	0.460E+00
51	0.317E+00
52	0.234E+00
53	0.888E-01
54	0.330E-01
55	0.186E-01
56	0.124E-01
57	0.908E-02
58	0.847E-02
59	0.564E+01
60	0.483E+01
61	0.416E+01
62	0.314E+01
63	0.243E+01
64	0.184E+01
65	0.144E+01
66	0.117E+01
67	0.965E+00
68	0.701E+00
69	0.536E+00
70	0.351E+00
71	0.252E+00
72	0.888E-01
73	0.318E-01
74	0.176E-01
75	0.117E-01
76	0.850E-02
77	0.401E+01
78	0.267E+01
79	0.228E+01
80	0.197E+01
81	0.148E+01
82	0.115E+01
83	0.867E+00
84	0.681E+00
85	0.550E+00
86	0.455E+00
87	0.331E+00
88	0.253E+00
89	0.165E+01
90	0.119E+01
91	0.419E-01
92	0.150E-01
93	0.834E-02
94	0.553E-02
95	0.403E-02

TABLE 3-11

Average Atmospheric Concentrations ($\mu g/m^3$), Including Dry Deposition Effects, Predicted by RTDMDEP

Receptor Number	RTDMDEP 30° Slope	RTDMDEP 45° Slope	RTDMDEP 60° Slope
1	0.396E+00	0.396E+00	0.396E+00
2	0.197E+01	0.197E+01	0.221E+01
3	0.238E+01	0.238E+01	0.238E+01
4	0.178E+01	0.178E+01	0.174E+01
5	0.661E+00	0.661E+00	0.661E+00
6	0.133E+01	0.133E+01	0.133E+01
7	0.140E+01	0.140E+01	0.140E+01
8	0.116E+01	0.105E+01	0.988E+00
9	0.174E+00	0.174E+00	0.174E+00
10	0.171E+00	0.174E+00	0.178E+00
11	0.169E+00	0.170E+00	0.171E+00
12	0.104E+00	0.106E+00	0.106E+00

travel time increases (i.e., distance from the source increases), the reduction in pollutant concentration due to depletion becomes apparent even at ground level.

3.5.3. Calculated Values of Pollutant Deposition

3.5.3.1. COMPDEP AND ISCST RESULTS -- Unlike atmospheric concentrations that were identical, predicted values for dry deposition made by COMPDEP (Table 3-12) are different than those of ISCST in flat terrain (Receptors 1 - 19). Deposition rates for receptors within 3000 meters of the source predicted by ISCST are on average 2.3 times higher than those of COMPDEP. Maximum deposition rates predicted by ISCST are as much as 10.0 times those of COMPDEP. Beyond this distance, however, the opposite is true as deposition rates predicted by COMPDEP are 1.1 to 4.2 times those of ISCST. Examination of the remaining receptors yields similar results. For points near the source, ISCST deposition rates vary from 1.0 to 37.6 and average 3.0 times those of COMPDEP. As downwind distance increases, values predicted by COMPDEP average 2.5 times higher than those of ISCST. Furthermore, for receptors 25, 50 and 100 meters above stack height, the point at which COMPDEP predictions exceed those of ISCST shifts toward the stack as terrain height increases. The fact that this trend is not seen for the receptors 300 meters above the stack is probably due to the increasing influence of the terrain correction factor used by COMPDEP.

TABLE 3-12

Average Annual Dry Deposition ($g/m^2/yr$) Predicted by COMPDEP and ISCST

Receptor Number	COMPDEP Deposition	ISCST Deposition
1	0.816E-03	0.814E-02
2	0.108E-01	0.626E-01
3	0.158E-01	0.782E-01
4	0.203E-01	0.872E-01
5	0.279E-01	0.956E-01
6	0.322E-01	0.917E-01
7	0.318E-01	0.703E-01
8	0.300E-01	0.562E-01
9	0.277E-01	0.454E-01
10	0.254E-01	0.371E-01
11	0.212E-01	0.256E-01
12	0.180E-01	0.185E-01
13	0.134E-01	0.105E-01
14	0.104E-01	0.693E-02
15	0.420E-02	0.188E-02
16	0.145E-02	0.475E-03
17	0.770E-03	0.225E-03
18	0.500E-03	0.139E-03
19	0.359E-03	0.983E-04
20	0.404E-02	0.150E+00
21	0.267E-01	0.335E+00
22	0.372E-01	0.331E+00
23	0.455E-01	0.297E+00
24	0.545E-01	0.217E+00
25	0.555E-01	0.156E+00
26	0.505E-01	0.981E-01
27	0.451E-01	0.699E-01
28	0.402E-01	0.522E-01
29	0.360E-01	0.404E-01
30	0.291E-01	0.257E-01
31	0.239E-01	0.177E-01
32	0.170E-01	0.962E-02
33	0.127E-01	0.611E-02
34	0.460E-02	0.156E-02
35	0.153E-02	0.402E-03
36	0.806E-03	0.197E-03
37	0.520E-03	0.125E-03
38	0.369E-03	0.899E-04
39	0.916E-01	0.344E+01
40	0.119E+00	0.908E+00
41	0.132E+00	0.574E+00
42	0.137E+00	0.394E+00
43	0.129E+00	0.215E+00
44	0.113E+00	0.134E+00
45	0.926E-01	0.787E-01
46	0.769E-01	0.533E-01
47	0.646E-01	0.383E-01

TABLE 3-12 (cont.)

Receptor Number	COMPDEP Deposition	ISCST Deposition
48	0.549E-01	0.287E-01
49	0.411E-01	0.173E-01
52	0.318E-01	0.115E-01
51	0.209E-01	0.613E-02
52	0.148E-01	0.386E-02
53	0.485E-02	0.105E-02
54	0.157E-02	0.307E-03
55	0.816E-03	0.161E-03
56	0.520E-03	0.108E-03
57	0.367E-03	0.798E-04
58	0.535E+00	0.344E+01
59	0.384E+00	0.908E+00
60	0.334E+00	0.574E+00
61	0.288E+00	0.394E+00
62	0.216E+00	0.215E+00
63	0.164E+00	0.134E+00
64	0.122E+00	0.787E-01
65	0.944E-01	0.533E-01
66	0.752E-01	0.383E-01
67	0.613E-01	0.287E-01
68	0.434E-01	0.173E-01
69	0.325E-01	0.115E-01
70	0.204E-01	0.613E-02
71	0.142E-01	0.386E-02
72	0.449E-02	0.105E-02
73	0.144E-02	0.307E-03
74	0.744E-03	0.161E-03
75	0.472E-03	0.108E-03
76	0.333E-03	0.798E-04
77	0.252E+00	0.344E+01
78	0.181E+00	0.908E+00
79	0.156E+00	0.574E+00
80	0.134E+00	0.394E+00
81	0.101E+00	0.215E+00
82	0.768E-01	0.134E+00
83	0.571E-01	0.787E-01
84	0.441E-01	0.533E-01
85	0.351E-01	0.383E-01
86	0.286E-01	0.287E-01
87	0.203E-01	0.173E-01
88	0.151E-01	0.115E-01
89	0.953E-02	0.613E-02
90	0.661E-02	0.386E-02
91	0.209E-02	0.105E-02
92	0.671E-03	0.307E-03
93	0.348E-03	0.161E-03
94	0.221E-03	0.108E-03
95	0.156E-03	0.798E-04

TABLE 3-13

Average Annual Dry Deposition ($g/m^2/yr$) Predicted by RTDMDEP, COMPDEP and ISCST

Receptor Number	RTDMDEP 30° Slope	RTDMDEP 45° Slope	RTDMDEP 60° Slope	COMPDEP	ISCST
1	0.382E-01	0.382E-01	0.382E-01	0.372E-01	0.331E+00
2	0.162E+00	0.162E+00	0.180E+00	0.132E+00	0.574E+00
3	0.179E+00	0.179E+00	0.179E+00	0.334E+00	0.574E+00
4	0.158E+00	0.158E+00	0.154E+00	0.156E+00	0.574E+00
5	0.584E-01	0.584E-01	0.584E-01	0.555E-01	0.156E+00
6	0.105E-01	0.105E-01	0.105E-01	0.113E+00	0.134E+00
7	0.973E-01	0.972E-01	0.972E-01	0.164E+00	0.134E+00
8	0.950E-01	0.836E-01	0.793E-01	0.768E-01	0.134E+00
9	0.129E-01	0.129E-01	0.129E-01	0.127E-01	0.611E-02
10	0.118E-01	0.120E-01	0.122E-01	0.148E-01	0.386E-02
11	0.104E-01	0.104E-01	0.105E-01	0.142E-01	0.386E-02
12	0.735E-02	0.745E-02	0.750E-02	0.661E-02	0.386E-02

3.5.3.2. RTDMDEP RESULTS -- Deposition rates were also calculated by RTDMDEP (Table 3-13) for terrain 25, 50, 100, and 300 meters above stack height. Differences in deposition rates predicted by the two models are less than for concentrations. COMPDEP predictions range from 0.8 to 1.9 times those of the RTDMDEP. Like COMPDEP, RTDMDEP predicted deposition rates 2.7 times lower on average and as much as 8.7 times lower than those of ISCST near the source. At larger downwind distances, RTDMDEP values were 1.9 to 3.1 times higher than those of ISCST.

3.5.4. Summary of Air Dispersion Modeling

Two complex terrain models, COMPDEP and RTDMDEP were developed to account for pollutant transport in complex terrain. Atmospheric deposition components were added to two existing, complex terrain models (U.S. EPA, 1986c), COMPLEX I and RTDM, so that the models could be used to estimate uptake of pollutants through the terrestrial food chain. Both COMPDEP and RTDMDEP contain algorithms taken from the MPTER-DS model (U.S. EPA, 1982a), which calculate deposition rates based on a dry deposition velocity and account for plume depletion due to deposition. Additional, state-of-the-art algorithms (CARB, 1986), were incorporated that automatically calculate particle size-dependent dry deposition velocities based on atmospheric and surface conditions. Finally, the capability to estimate wet deposition was incorporated into these models through the use of scavenging coefficients.

To evaluate the impact of the new models' treatment of complex terrain and deposition, comparisons were made between their results and those of the widely used ISCST model. Comparisons between predicted concentrations showed, on the average, ISCST generally estimated marginally higher annual average atmospheric pollutant concentrations that were roughly twice those of COMPDEP and three times those of RTDMDEP. A few isolated cases existed in which ISCST actually gave slightly unconservative predictions with respect to those made by COMPDEP. These occurred primarily at large downwind distances and at receptor elevations of 100.0 meters above stack elevation. Many risk assessments, such as those weighted according to population density, are more a function of overall average pollutant concentration near the source than a maximum concentration at a single point. Based on the results of this analysis, use of ISCST in these risk assessments would be expected to give results that would range from about 0.9 to 3.0 times those obtained if either COMPDEP or RTDMDEP were used. Hence, for such assessments, use of ISCST in complex terrain appears to be valid.

Somewhat different conclusion can be drawn when considering maximum point concentrations. Whereas differences in average concentrations predicted by flat and complex terrain models were small, differences in maximum point concentrations were much more significant, differing in some instances by an order of magnitude. In all cases, the maximum point concentrations were predicted by ISCST. Thus in some cases, such as for risk assessments that base their exposures on maximum point concentrations of pollutant, use of ISCST in complex terrain must be questioned. If the user can tolerate a factor of ten conservatism, this model can be applied.

However, if this is not the case, consideration should be given to either COMPDEP or RTDMDEP. Based on the experience gained during this study, COMPDEP is recommended over RTDMDEP for use in these situations. It is easier to use, faster, and can be applied to more physical scenarios than RTDMDEP. Only for cases in which concentrations estimated by COMPDEP appear to be too conservative is RTDMEP recommended for use.

It should be noted that in spite of their technical improvements, currently recommended complex terrain screening models (U.S. EPA, 1986c) are still rather limited in their ability to model atmospheric transport in complex terrain. Being of their Gaussian plume formulation, their treatment of terrain is limited to a few algorithms that modify plume height. In the future it is possible that a refined complex terrain model, capable of realistically simulating flow in complex terrain, will be developed that can be used to obtain estimates of annual average atmospheric pollutant concentrations.

Comparisons between predicted deposition rates showed that those predicted COMPDEP and RTDMDEP were are as much as 39 times lower than those of ISCST. The deposition algorithms incorporated into COMPDEP and RTDMDEP are, in theory, more advanced than those contained in ISCST and should provide more realistic estimates of pollutant deposition rate. In addition unlike ISCST, which require the user input highly subjective values for reflection coefficient, they offer a systematic and automated means by which deposition velocities can be estimated. However, accurate conclusions regarding which of these algorithms provides the best estimates of dry deposition velocity cannot be obtained without reference to quantitative evaluations between predicted and measured values. Unfortunately such evaluations are rare. Bowers and Anderson (1981) found that ISCST produced results within a factor of two of measured values. In some instances calculated values were lower than measured values. Based on these results, deposition rates predicted by COMPDEP and RTDMDEP appear to grossly underestimate pollutant deposition rates. However, these deposition velocities were calculated under a limited set of atmospheric conditions and downwind distances and for particle sizes that ranged from 50 to 200 microns, much larger than those addressed in this analysis. In contrast to these results Tesche et al. (1987) indicated that under certain conditions ISCST can significantly overestimate dry deposition. Although no quantitative estimates of the degree of overprediction by ISCST were reported, the Tesche et al. (1987) results indicate that COMPDEP and RTDMDEP may provide better estimates of deposition rate under certain conditions. Thus, current evaluations of ISCST performance in estimating deposition rate appear contradictory. Until more detailed evaluations of the performance of models such as ISCST in estimating deposition rates are available, meaningful recommendations regarding whether the deposition algorithms developed for this study are indeed superior to those found in ISCST cannot be made.

3.6. DETERMINATION OF AREAL-AVERAGE DEPOSITION

In this document, deposition values were derived from the COMPDEP model described in Chapter 3 using unit (1 g/s) emission rate. The MWC modeled was a hypothetical facility in Western Florida. Both wet and dry deposition was determined. These values can be used to determine annual average deposition for various areas (areal averages) surrounding the combustor. These areas are defined as concentric rings of varying radii with the source at the center. The maximum radius is 50 km surrounding the combustion facility. Appendix A illustrates the concentric rings and lists the receptors within each radius. In this document, an annual average deposition value is calculated for each of the three exposure scenarios. As discussed in Chapter 2, a hypothetical MWC will be used as the combustion facility and examples shown for cadmium and benzo(a)pyrene. The output for the COMPDEP model, run at unit emission rates for cadmium and benzo(a)pyrene is shown in Tables 3-14 - 3-18.

To support development of regulations for MWCs under the Clean Air Act, Section 111, pollutant emission rates representative of a variety of facility types were estimated based on available MWC emission test reports (Myers, 1989). Myers (1989) estimated these rates from both measured particulate matter and PCDD/PCDF emissions and developed metal to particulate ratios and organic compound to PCDD/PCDF ratios so that the emission rates of specific metals and organic compounds could be quantified. Emission rate estimates for cadmium and benzo(a)pyrene have been selected from this source for use in developing the three exposure scenarios (A, B and C) in this document. The selections are somewhat arbitrary and are chosen primarily to illustrate the range in emission rates observed for facilities of different designs and sizes.

TABLE 3-14

Annual Dry Deposition for Benzo(a)Pyrene
(g/m^2)

Rc	Dp	Rc	Dp	Rc	Dp	Rc	Dp
1	0.3406E-02	2	0.1726E-02	3	0.2525E-02	4	0.6839E-02
5	0.3870E-02	6	0.3619E-02	7	0.3240E-02	8	0.2324E-02
9	0.2259E-02	10	0.4180E-02	11	0.5416E-02	12	0.6874E-02*
13	0.4720E-02	14	0.3602E-02	15	0.3048E-02	16	0.2686E-02
17	0.1816E-02	18	0.1098E-02	19	0.1607E-02	20	0.3914E-02
21	0.2019E-02	22	0.1895E-02	23	0.1752E-02	24	0.1290E-02
25	0.1158E-02	26	0.2197E-02	27	0.3081E-02	28	0.3806E-02
29	0.2580E-02	30	0.1988E-02	31	0.1662E-02	32	0.1477E-02
33	0.1340E-02	34	0.1036E-02	35	0.1534E-02	36	0.3166E-02
37	0.1320E-02	38	0.1142E-02	39	0.1072E-02	40	0.8627E-03
41	0.7473E-03	42	0.1410E-02	43	0.2140E-02	44	0.2609E-02
45	0.1765E-02	46	0.1497E-02	47	0.1276E-02	48	0.1082E-02
49	0.1301E-02	50	0.9607E-03	51	0.1396E-02	52	0.3014E-02
53	0.1201E-02	54	0.1094E-02	55	0.1008E-02	56	0.8650E-03
57	0.7574E-03	58	0.1378E-02	59	0.2078E-02	60	0.2553E-02
61	0.1702E-02	62	0.1491E-02	63	0.1267E-02	64	0.1046E-02
65	0.6974E-03	66	0.4789E-03	67	0.6448E-03	68	0.1505E-02
69	0.6950E-03	70	0.7882E-03	71	0.7073E-03	72	0.6547E-03
73	0.5009E-03	74	0.9432E-03	75	0.1495E-02	76	0.1845E-02
77	0.1137E-02	78	0.9431E-03	79	0.7631E-03	80	0.6420E-03
81	0.3530E-03	82	0.2605E-03	83	0.3291E-03	84	0.7448E-03
85	0.3812E-03	86	0.5065E-03	87	0.4590E-03	88	0.4476E-03
89	0.2947E-03	90	0.5883E-03	91	0.9935E-03	92	0.1212E-02
93	0.7194E-03	94	0.5754E-03	95	0.4446E-03	96	0.3748E-03
97	0.1530E-03	98	0.1188E-03	99	0.1455E-03	100	0.3134E-03
101	0.1687E-03	102	0.2372E-03	103	0.2188E-03	104	0.2220E-03
105	0.1409E-03	106	0.2901E-03	107	0.5186E-03	108	0.6221E-03
109	0.3532E-03	110	0.2766E-03	111	0.2147E-03	112	0.1785E-03
113	0.8797E-04	114	0.6839E-04	115	0.8325E-04	116	0.1769E-03
117	0.9512E-04	118	0.1329E-03	119	0.1223E-03	120	0.1274E-03
121	0.8170E-04	122	0.1672E-03	123	0.3037E-03	124	0.3628E-03
125	0.2018E-03	126	0.1589E-03	127	0.1260E-03	128	0.1035E-03
129	0.5844E-04	130	0.4589E-04	131	0.5633E-04	132	0.1181E-03
133	0.6248E-04	134	0.8775E-04	135	0.8048E-04	136	0.8491E-04
137	0.5361E-04	138	0.1090E-03	139	0.1986E-03	140	0.2375E-03
141	0.1320E-03	142	0.1054E-03	143	0.8364E-04	144	0.6862E-04
145	0.4233E-04	146	0.3358E-04	147	0.4182E-04	148	0.8662E-04
149	0.4486E-04	150	0.6344E-04	151	0.5821E-04	152	0.6166E-04
153	0.3832E-04	154	0.7762E-04	155	0.1416E-03	156	0.1696E-03
157	0.9480E-04	158	0.7645E-04	159	0.6036E-04	160	0.4965E-04

* Maximum annual dry deposition = 0.6874E-02 at receptor: 12

Rc = Receptor
Dp = Deposition

TABLE 3-15

Annual Dry Deposition for Cadmium
(g/m^2)

Rc	Dp	Rc	Dp	Rc	Dp	Rc	Dp
1	0.7892E-02	2	0.3955E-02	3	0.5800E-02	4	0.1579E-01*
5	0.8985E-02	6	0.8406E-02	7	0.7526E-02	8	0.5376E-02
9	0.5246E-02	10	0.9643E-02	11	0.1234E-01	12	0.1564E-01
13	0.1076E-01	14	0.8257E-02	15	0.6981E-02	16	0.6188E-02
17	0.4195E-02	18	0.2496E-02	19	0.3667E-02	20	0.9011E-02
21	0.4652E-02	22	0.4326E-02	23	0.3996E-02	24	0.2918E-02
25	0.2657E-02	26	0.4996E-02	27	0.6899E-02	28	0.8537E-02
29	0.5787E-02	30	0.4496E-02	31	0.3772E-02	32	0.3365E-02
33	0.3091E-02	34	0.2359E-02	35	0.3503E-02	36	0.7273E-02
37	0.3035E-02	38	0.2603E-02	39	0.2442E-02	40	0.1954E-02
41	0.1717E-02	42	0.3210E-02	43	0.4813E-02	44	0.5876E-02
45	0.3979E-02	46	0.3397E-02	47	0.2904E-02	48	0.2466E-02
49	0.2995E-02	50	0.2189E-02	51	0.3185E-02	52	0.6913E-02
53	0.2763E-02	54	0.2513E-02	55	0.2314E-02	56	0.1981E-02
57	0.1750E-02	58	0.3165E-02	59	0.4726E-02	60	0.5809E-02
61	0.3868E-02	62	0.3403E-02	63	0.2896E-02	64	0.2391E-02
65	0.1600E-02	66	0.1091E-02	67	0.1469E-02	68	0.3442E-02
69	0.1597E-02	70	0.1818E-02	71	0.1633E-02	72	0.1513E-02
73	0.1160E-02	74	0.2176E-02	75	0.3422E-02	76	0.4223E-02
77	0.2598E-02	78	0.2158E-02	79	0.1745E-02	80	0.1469E-02
81	0.8043E-03	82	0.5933E-03	83	0.7493E-03	84	0.1695E-02
85	0.8723E-03	86	0.1171E-02	87	0.1065E-02	88	0.1041E-02
89	0.6812E-03	90	0.1359E-02	91	0.2282E-02	92	0.2781E-02
93	0.1648E-02	94	0.1318E-02	95	0.1014E-02	96	0.8559E-03
97	0.3423E-03	98	0.2634E-03	99	0.3225E-03	100	0.6965E-03
101	0.3758E-03	102	0.5290E-03	103	0.4903E-03	104	0.4993E-03
105	0.3195E-03	106	0.6552E-03	107	0.1163E-02	108	0.1394E-02
109	0.7864E-03	110	0.6160E-03	111	0.4795E-03	112	0.3977E-03
113	0.1931E-03	114	0.1467E-03	115	0.1784E-03	116	0.3826E-03
117	0.2052E-03	118	0.2832E-03	119	0.2611E-03	120	0.2742E-03
121	0.1807E-03	122	0.3652E-03	123	0.6565E-03	124	0.7840E-03
125	0.4315E-03	126	0.3410E-03	127	0.2737E-03	128	0.2236E-03
129	0.1260E-03	130	0.9588E-04	131	0.1177E-03	132	0.2503E-03
133	0.1310E-03	134	0.1801E-03	135	0.1648E-03	136	0.1760E-03
137	0.1155E-03	138	0.2301E-03	139	0.4125E-03	140	0.4939E-03
141	0.2715E-03	142	0.2188E-03	143	0.1768E-03	144	0.1440E-03
145	0.8990E-04	146	0.6866E-04	147	0.8576E-04	148	0.1807E-03
149	0.9179E-04	150	0.1262E-03	151	0.1152E-03	152	0.1237E-03
153	0.8065E-04	154	0.1587E-03	155	0.2834E-03	156	0.3406E-03
157	0.1886E-03	158	0.1545E-03	159	0.1245E-03	160	0.1016E-03

* Maximum annual dry deposition = 0.1579E-01 at receptor: 4

Rc = Receptor
Dp = Deposition

TABLE 3-16

Annual Wet Deposition for Benzo(a)Pyrene
(g/m^2)

Rc	Dp	Rc	Dp	Rc	Dp	Rc	Dp
1	0.7370E-01	2	0.5439E-01	3	0.4516E-01	4	0.6277E-01
5	0.5229E-01	6	0.4445E-01	7	0.4787E-01	8	0.5710E-01
9	0.7245E-01	10	0.8588E-01	11	0.7578E-01	12	0.1129E+00*
13	0.8744E-01	14	0.8513E-01	15	0.1021E+00	16	0.1123E+00
17	0.2208E-01	18	0.1657E-01	19	0.1385E-01	20	0.2098E-01
21	0.1690E-01	22	0.1364E-01	23	0.1564E-01	24	0.1836E-01
25	0.2147E-01	26	0.2555E-01	27	0.2185E-01	28	0.3545E-01
29	0.2477E-01	30	0.2434E-01	31	0.3133E-01	32	0.3634E-01
33	0.8540E-02	34	0.5868E-02	35	0.5195E-02	36	0.8787E-02
37	0.7105E-02	38	0.5249E-02	39	0.6497E-02	40	0.7178E-02
41	0.7742E-02	42	0.9756E-02	43	0.7707E-02	44	0.1409E-01
45	0.8865E-02	46	0.8847E-02	47	0.1248E-01	48	0.1410E-01
49	0.3401E-02	50	0.2376E-02	51	0.2106E-02	52	0.3681E-02
53	0.3001E-02	54	0.2017E-02	55	0.2774E-02	56	0.2950E-02
57	0.2857E-02	58	0.3874E-02	59	0.3023E-02	60	0.5958E-02
61	0.3344E-02	62	0.3414E-02	63	0.5037E-02	64	0.5715E-02
65	0.1041E-02	66	0.7418E-03	67	0.6586E-03	68	0.1140E-02
69	0.9100E-03	70	0.5973E-03	71	0.8922E-03	72	0.9517E-03
73	0.8035E-03	74	0.1241E-02	75	0.9495E-03	76	0.1895E-02
77	0.1015E-02	78	0.1014E-02	79	0.1525E-02	80	0.1744E-02
81	0.3906E-03	82	0.2730E-03	83	0.2397E-03	84	0.4250E-03
85	0.3223E-03	86	0.2196E-03	87	0.3333E-03	88	0.3688E-03
89	0.2899E-03	90	0.4795E-03	91	0.3545E-03	92	0.6974E-03
93	0.3751E-03	94	0.3651E-03	95	0.5593E-03	96	0.6437E-03
97	0.1220E-03	98	0.8024E-04	99	0.6800E-04	100	0.1321E-03
101	0.9150E-04	102	0.6763E-04	103	0.9911E-04	104	0.1154E-03
105	0.8766E-04	106	0.1498E-03	107	0.1058E-03	108	0.2047E-03
109	0.1119E-03	110	0.1054E-03	111	0.1678E-03	112	0.1959E-03
113	0.5349E-04	114	0.3333E-04	115	0.2740E-04	116	0.5877E-04
117	0.3787E-04	118	0.2967E-04	119	0.4161E-04	120	0.4945E-04
121	0.3739E-04	122	0.6412E-04	123	0.4403E-04	124	0.8529E-04
125	0.4720E-04	126	0.4320E-04	127	0.7128E-04	128	0.8473E-04
129	0.2714E-04	130	0.1614E-04	131	0.1302E-04	132	0.3055E-04
133	0.1862E-04	134	0.1515E-04	135	0.2050E-04	136	0.2431E-04
137	0.1839E-04	138	0.3153E-04	139	0.2117E-04	140	0.4154E-04
141	0.2322E-04	142	0.2066E-04	143	0.3517E-04	144	0.4268E-04
145	0.1497E-04	146	0.8555E-05	147	0.6832E-05	148	0.1734E-04
149	0.1013E-04	150	0.8459E-05	151	0.1112E-04	152	0.1296E-04
153	0.9818E-05	154	0.1685E-04	155	0.1108E-04	156	0.2217E-04
157	0.1250E-04	158	0.1079E-04	159	0.1892E-04	160	0.2348E-04

* Maximum annual wet deposition = 0.1129E+00 at receptor: 12

Rc = Receptor
Dp = Deposition

TABLE 3-17

Annual Wet Deposition for Cadmium
(g/m^2)

Rc	Dp	Rc	Dp	Rc	Dp	Rc	Dp
1	0.1202E+00	2	0.8744E-01	3	0.7260E-01	4	0.1039E+00
5	0.8496E-01	6	0.7355E-01	7	0.8140E-01	8	0.9575E-01
9	0.1213E+00	10	0.1447E+00	11	0.1285E+00	12	0.1895E+00*
13	0.1460E+00	14	0.1395E+00	15	0.1684E+00	16	0.1861E+00
17	0.3159E-01	18	0.2292E-01	19	0.1903E-01	20	0.2982E-01
21	0.2330E-01	22	0.1958E-01	23	0.2260E-01	24	0.2659E-01
25	0.3100E-01	26	0.3703E-01	27	0.3171E-01	28	0.5080E-01
29	0.3549E-01	30	0.3407E-01	31	0.4478E-01	32	0.5262E-01
33	0.1027E-01	34	0.6933E-02	35	0.6011E-02	36	0.1038E-01
37	0.8183E-02	38	0.6311E-02	39	0.7662E-02	40	0.8549E-02
41	0.9276E-02	42	0.1157E-01	43	0.9147E-02	44	0.1654E-01
45	0.1051E-01	46	0.1028E-01	47	0.1482E-01	48	0.1703E-01
49	0.3406E-02	50	0.2368E-02	51	0.2086E-02	52	0.3669E-02
53	0.2978E-02	54	0.2030E-02	55	0.2749E-02	56	0.2917E-02
57	0.2843E-02	58	0.3813E-02	59	0.2976E-02	60	0.5883E-02
61	0.3311E-02	62	0.3357E-02	63	0.5013E-02	64	0.5759E-02
65	0.9640E-03	66	0.6904E-03	67	0.6152E-03	68	0.1058E-02
69	0.8463E-03	70	0.5515E-03	71	0.8285E-03	72	0.8828E-03
73	0.7404E-03	74	0.1150E-02	75	0.8816E-03	76	0.1763E-02
77	0.9409E-03	78	0.9428E-03	79	0.1418E-02	80	0.1621E-02
81	0.3619E-03	82	0.2535E-03	83	0.2229E-03	84	0.3950E-03
85	0.3003E-03	86	0.2035E-03	87	0.3107E-03	88	0.3430E-03
89	0.2685E-03	90	0.4459E-03	91	0.3294E-03	92	0.6482E-03
93	0.3485E-03	94	0.3388E-03	95	0.5197E-03	96	0.5955E-03
97	0.1141E-03	98	0.7528E-04	99	0.6366E-04	100	0.1242E-03
101	0.8632E-04	102	0.6357E-04	103	0.9366E-04	104	0.1085E-03
105	0.8240E-04	106	0.1409E-03	107	0.9943E-04	108	0.1924E-03
109	0.1053E-03	110	0.9878E-04	111	0.1573E-03	112	0.1832E-03
113	0.5067E-04	114	0.3168E-04	115	0.2600E-04	116	0.5587E-04
117	0.3612E-04	118	0.2821E-04	119	0.3980E-04	120	0.4712E-04
121	0.3562E-04	122	0.6109E-04	123	0.4195E-04	124	0.8131E-04
125	0.4500E-04	126	0.4112E-04	127	0.6762E-04	128	0.8037E-04
129	0.2605E-04	130	0.1555E-04	131	0.1253E-04	132	0.2934E-04
133	0.1794E-04	134	0.1456E-04	135	0.1983E-04	136	0.2349E-04
137	0.1775E-04	138	0.3046E-04	139	0.2047E-04	140	0.4015E-04
141	0.2241E-04	142	0.1996E-04	143	0.3379E-04	144	0.4103E-04
145	0.1456E-04	146	0.8351E-05	147	0.6671E-05	148	0.1682E-04
149	0.9859E-05	150	0.8212E-05	151	0.1087E-04	152	0.1270E-04
153	0.9593E-05	154	0.1650E-04	155	0.1087E-04	156	0.2171E-04
157	0.1222E-04	158	0.1058E-04	159	0.1841E-04	160	0.2285E-04

* Maximum annual wet deposition = 0.1895E+00 at receptor: 12

Rc = Receptor
Dp = Deposition

TABLE 3-18

Average Hourly Concentrations for Benzo(a)Pyrene ($\mu g/m^3$)

Rc	Dp	Rc	Dp	Rc	Dp	Rc	Dp
1	0.4043E-01	2	0.2242E-01	3	0.3095E-01	4	0.7807E-01
5	0.4649E-01	6	0.4867E-01	7	0.4430E-01	8	0.3720E-01
9	0.3550E-01	10	0.6131E-01	11	0.7950E-01	12	0.9888E-01*
13	0.6635E-01	14	0.5002E-01	15	0.4320E-01	16	0.3588E-01
17	0.2137E-01	18	0.1418E-01	19	0.1969E-01	20	0.4450E-01
21	0.2421E-01	22	0.2561E-01	23	0.2413E-01	24	0.2040E-01
25	0.1782E-01	26	0.3224E-01	27	0.4574E-01	28	0.5532E-01
29	0.3633E-01	30	0.2745E-01	31	0.2321E-01	32	0.1961E-01
33	0.1616E-01	34	0.1339E-01	35	0.1905E-01	36	0.3688E-01
37	0.1590E-01	38	0.1530E-01	39	0.1463E-01	40	0.1323E-01
41	0.1107E-01	42	0.2031E-01	43	0.3109E-01	44	0.3721E-01
45	0.2429E-01	46	0.2027E-01	47	0.1740E-01	48	0.1422E-01
49	0.1635E-01	50	0.1275E-01	51	0.1790E-01	52	0.3646E-01
53	0.1484E-01	54	0.1449E-01	55	0.1356E-01	56	0.1287E-01
57	0.1114E-01	58	0.1950E-01	59	0.2922E-01	60	0.3534E-01
61	0.2309E-01	62	0.2015E-01	63	0.1739E-01	64	0.1403E-01
65	0.9254E-02	66	0.6497E-02	67	0.8487E-02	68	0.1893E-01
69	0.8919E-02	70	0.1034E-01	71	0.9373E-02	72	0.9370E-02
73	0.7522E-02	74	0.1338E-01	75	0.2066E-01	76	0.2531E-01
77	0.1531E-01	78	0.1282E-01	79	0.1080E-01	80	0.8942E-02
81	0.4939E-02	82	0.3527E-02	83	0.4334E-02	84	0.9690E-02
85	0.5016E-02	86	0.6477E-02	87	0.5826E-02	88	0.6056E-02
89	0.4488E-02	90	0.8251E-02	91	0.1343E-01	92	0.1637E-01
93	0.9487E-02	94	0.7753E-02	95	0.6379E-02	96	0.5267E-02
97	0.2341E-02	98	0.1794E-02	99	0.2139E-02	100	0.4550E-02
101	0.2503E-02	102	0.3536E-02	103	0.3210E-02	104	0.3444E-02
105	0.2335E-02	106	0.4494E-02	107	0.7786E-02	108	0.9345E-02
109	0.5248E-02	110	0.4196E-02	111	0.3358E-02	112	0.2753E-02
113	0.1458E-02	114	0.1164E-02	115	0.1386E-02	116	0.2852E-02
117	0.1596E-02	118	0.2348E-02	119	0.2146E-02	120	0.2334E-02
121	0.1509E-02	122	0.2974E-02	123	0.5295E-02	124	0.6307E-02
125	0.3494E-02	126	0.2776E-02	127	0.2189E-02	128	0.1790E-02
129	0.1035E-02	130	0.8489E-03	131	0.1016E-02	132	0.2044E-02
133	0.1151E-02	134	0.1734E-02	135	0.1596E-02	136	0.1748E-02
137	0.1094E-02	138	0.2192E-02	139	0.3969E-02	140	0.4704E-02
141	0.2584E-02	142	0.2045E-02	143	0.1598E-02	144	0.1305E-02
145	0.7901E-03	146	0.6605E-03	147	0.7956E-03	148	0.1575E-02
149	0.8893E-03	150	0.1361E-02	151	0.1260E-02	152	0.1387E-02
153	0.8482E-03	154	0.1720E-02	155	0.3149E-02	156	0.3719E-02
157	0.2030E-02	158	0.1603E-02	159	0.1245E-02	160	0.1016E-02

* Maximum average hourly concentration = 0.9888E-01 at receptor: 12

Rc = Receptor
Dp = Deposition

Cadmium emissions were found to be related to both facility type and particulate matter (PM) emission rate. Highest emissions per unit PM emission rate (5539 μg Cd/g PM) were for mass burn (MB) facilities with PM control but without acid gas control, where as lowest rates were for refuse-derived fuel (RDF) combustors with only PM control (395 μg Cd/g PM) or for all combustors with acid gas control (390 μg Cd/g PM).

Benzo(a)pyrene emissions were assumed to be related to total PCDD/PCDF concentrations, in spite of the lack of an observed correlation between these rates. The ratio of B(a)P:PCDD/PCDF was assumed to be 11.15 for all MWC facility types. Based on the emission rate ratios of both cadmium to particulate matter and benzo(a)pyrene to PCDD/PCDF, Myers (1989) modeled representative pollutant emission rate estimates for a variety of facility types. Cadmium emissions for a large MB facility were estimated to be 1843 kg Cd/yr and will be used to estimate deposition. Since benzo(a)pyrene emissions were assumed proportional to PCDD/PCDF emissions, and since the latter differ from particulates in emission characteristics, the type of facility selected to represent the various cases will differ for these two chemicals. The highest benzo(a)pyrene emissions (6.5 kg B(a)P/yr) were chosen from a MB facility. Table 3-19 lists the emission estimated for the different facility types.

The modeled wet and dry deposition rates (from Chapter 3) were adjusted by the above emission rates, and areal-average deposition values determined for the three exposure scenarios (See Table 3-20). Areal-average deposition was determined for Scenario B by

TABLE 3-19

Emission Rates (kg/yr) for Planned MWCs

Plant Type	Particulate Matter[a]	Cadmium[b]	PCDD/ PCDF[a,c]	Benzo(a)-Pyrene[b]
Small MB/WW	29610.36	164.0	10.00050	0.36
Medium MB/WW	118386.99	655.75	0.00323	2.31
Large MB/WW	332708.26	1842.87	0.00909	6.5
MB/Ref	73935.17	409.53	0.00303	2.17
MB/RC	155127.78	859.25	0.00633	4.53
RDF	364232.77	143.87	0.00996	7.13
RDF (Cofired)	322956.08	63.78	0.00883	6.32
MI/EA	56825.76	314.76	0.00097	0.69
MI/SA No heat rec	9071.80	12.05	0.00016	0.14
MI/SA	18869.34	35.64	0.00061	0.43
Bubbling Bed FBC	33000.00	13.04	0.00045	0.32
Circulating Bed FBC	33000.00	6.52	0.00895	6.41

[a] Measured emissions

[b] Predicted emissions

[c] Particulate emission is expressed as TCDD equivalents. TCDD equivalents for MWC with PM control were determined by multiplying the emission rate by the factor 0.0156.

TABLE 3-20

Summary of Deposition Rates for Cadmium and Benzo(a)Pyrene

	A			B			C		
	Unit Emission ($g/m^2/yr$)	Emission Rate (g/s)	Adjusted Deposition ($g/m^2/yr$)	Unit Emission ($g/m^2/yr$)	Emission Rate (g/s)	Adjusted Deposition ($g/m^2/yr$)	Unit Emission ($g/m^2/yr$)	Emission Rate (g/s)	Adjusted Deposition ($g/m^2/yr$)
B(a)P:									
Dyd	1.83×10^{-4}	2.06×10^{-4}	3.77×10^{-8}	1.19×10^{-3}	2.06×10^{-4}	2.45×10^{-7}	6.87×10^{-3}	2.06×10^{-4}	1.42×10^{-6}
Dyw	1.33×10^{-4}	2.06×10^{-4}	2.74×10^{-8}	3.27×10^{-3}	2.06×10^{-4}	6.74×10^{-7}	1.13×10^{-1}	2.06×10^{-4}	2.33×10^{-5}
Cy	2.96×10^{-3}*	2.06×10^{-4}	6.10×10^{-7}*	1.61×10^{-2}*	2.06×10^{-4}	3.32×10^{-6}*	9.8×10^{-2}*	2.06×10^{-4}	2.02×10^{-5}*
Cd:									
Dyd	4.03×10^{-4}	.058	2.34×10^{-5}	2.74×10^{-3}	.058	1.59×10^{-4}	1.58×10^{-2}	.058	9.1×10^{-4}
Dyw	1.14×10^{-4}	.058	6.6×10^{-6}	3.74×10^{-3}	.058	2.17×10^{-4}	1.90×10^{-1}	.058	1.1×10^{-2}

*Cy = ambient air concentration of pollutant ($\mu g/m^3$)

using deposition for radii ≤5 km, whereas Scenario A was calculated from the radii ≤50 km. The point of maximal deposition (at 0.2 km) was used for Scenario C. These values are the initial input for the determination of soil and water concentrations. Appendix A shows the formula for calculation of the areal-average deposition.

4. CALCULATING SOIL CONCENTRATION

4.1. INTRODUCTION

Contaminants associated with combustor emissions are subject to deposition on soil surfaces in the vicinity of the point source. Following deposition, contaminants may be incorporated into the upper layers of soil where crops or other vegetation are grown. This chapter focuses on the estimation of the cumulative concentration of a given contaminant in the soil following deposition from a combustor. The concentrations will be used to estimate the risk to humans who may ingest soil directly or consume vegetation and animals that have been exposed to contaminated soil.

Contaminant levels in soil are the result of both wet and dry particulate deposition onto soil and the loss of contaminant due to leaching, abiotic and biotic degradation and volatilization. Determination of annual deposition was discussed in Chapter 3. Figure 4-1 illustrates the number of factors required to determine the cumulative soil concentration. Soil conditions such as pH, soil structure and characteristics, organic matter content and water content affect the distribution and mobility of contaminants after deposition onto the soil. Loss of contaminants from the soil is modeled using rates that depend on site-specific data about the physical and chemical characteristics of the soil.

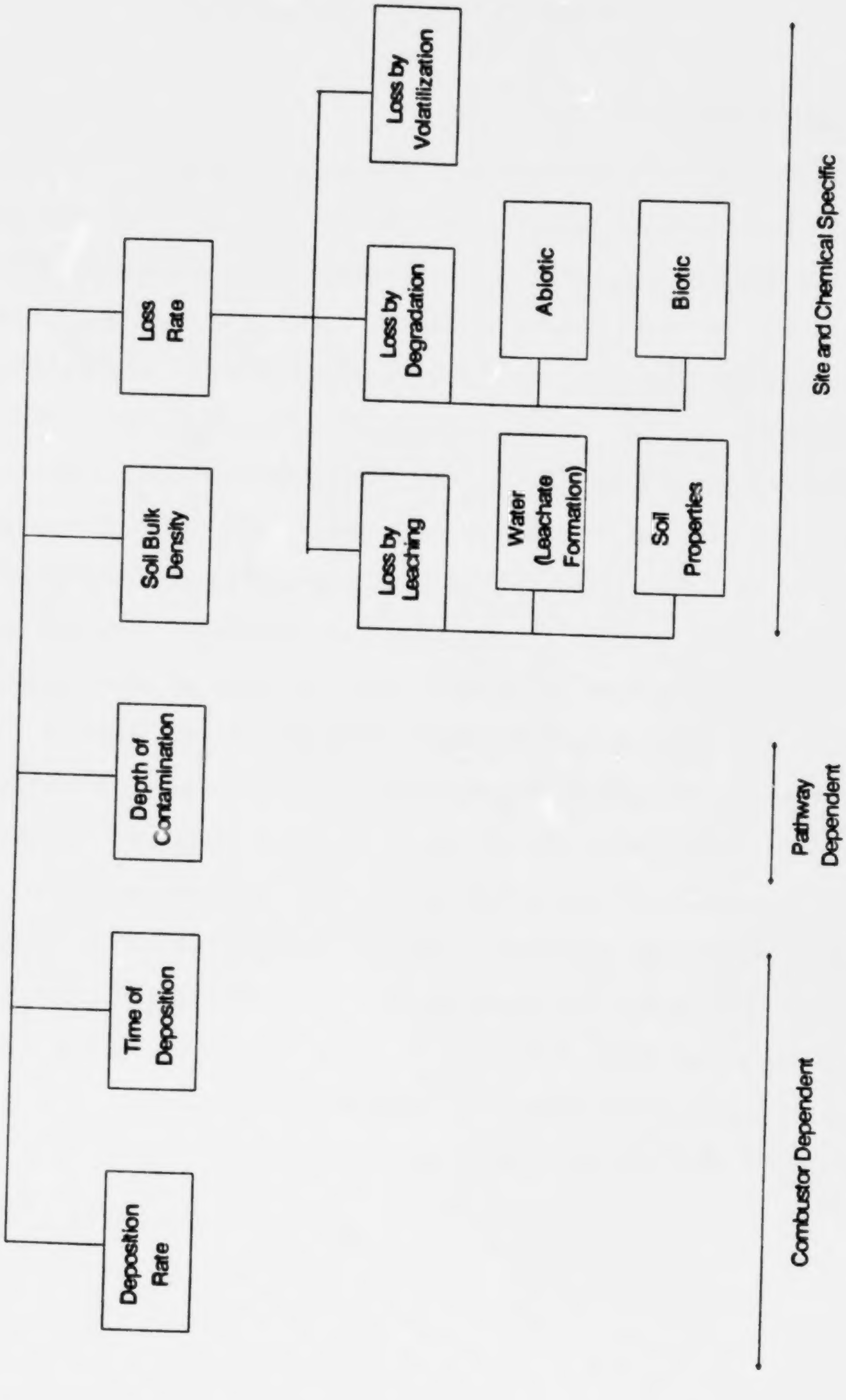
Deposition Rate
Time of Deposition
Depth of Contamination
Soil Bulk Density
Loss Rate
Loss by Leaching
Water (Leachate Formation)
Soil Properties
Loss by Degradation
Abiotic
Biotic
Loss by Volatilization
Combustor Dependent
Pathway Dependent
Site and Chemical Specific

Figure 4-1. Determination of Cumulative Soil Concentration

4.2. OVERVIEW OF SOIL CONCENTRATION CALCULATIONS

The cumulative soil concentration of a pollutant is derived from the wet and dry deposition rates over the lifetime of the combustor and the contaminant loss rate from the soil. Contaminants may be lost from soils as a result of numerous factors, including leaching, abiotic and biotic degradation and volatilization. The equations used to estimate contaminant concentrations and losses in soil are adapted from Travis et al., (1983). A summary of the general assumptions for the soil concentration estimates is presented in Table 4-1.

4.2.1. Calculating Cumulative Soil Concentration. The cumulative concentration of a pollutant (mg pollutant/g soil) is calculated from the wet and dry deposition rates and the loss constant:

$$Sc = \frac{(Dyd + Dyw) \cdot [1.0 - \exp(-ks \cdot Tc)] \cdot 0.1}{Z \cdot BD \cdot ks}$$

(Equation 4-1)

where:

Sc = soil concentration of pollutant after total time period of deposition (mg pollutant/g soil)
Dyd = yearly dry deposition rate of pollutant (g pollutant/m^2/yr)
Dyw = yearly wet deposition rate of pollutant (g pollutant/m^2/yr)
ks = soil loss constant (yr^{-1})
Tc = total time period over which deposition occurs (yrs)
0.1 = units conversion factor
Z = soil depth (cm)
BD = soil bulk density (g/cm^3)

TABLE 4-1

Assumptions for Soil Concentration Calculations

Assumptions	Ramifications/Limitations
If soil incorporation of contaminant is assumed, incorporation depth is 20 cm, and the upper 20 cm soil layer has a dry mass of 2.7 x 10^3 Mg/ha.	By this assumption, a soil concentration of 1 μg/g corresponds to a pollutant application of 2.7 kg/ha. If actual depth (and mass) is less, contaminant concentration could be underpredicted (and vice versa).
Soil background concentration is not considered.	The calculated soil concentration is only the soil increment due to combustion fallout, and could underpredict the soil concentration.
Degradation of organic contaminants is first-order.	Could over- or underpredict degradation rate, which is complex and not necessarily first-order.
Trace metal contaminants are assumed to be conserved indefinitely in the upper layer of soil unless loss constants are available.	Although most heavy metals are tightly bound to the soil, measurements show that this assumption often overpredicts concentrations (i.e., at low pH metals become increasingly mobile and may be leached out of the upper soil layers).
Erosion is not considered as a loss mechanism.	Could overestimate the soil contaminant concentration because significant losses can occur by this process.
The fact that loss processes occur simultaneously is not taken into account.	Could overpredict losses because contaminant concentration available for each loss process would be overpredicted.

The cumulative soil concentration, Sc, represents the concentration increment due to accumulation of contaminant deposited onto soil from the combustor. The cumulative soil concentration does not represent the total concentration, because it does not take into account background concentrations of the contaminant that may already be present, whether natural or from other pollution sources.

4.2.2. Calculating the Soil Loss Constant. The soil loss constant is determined from the loss due to leaching and the loss due to degradative and volatilization processes:

$$ks = ksl + ksg + ksv \qquad \text{(Equation 4-2)}$$

where:

ks = soil loss constant due to all processes (yr^{-1})
ksl = loss constant due to leaching (yr^{-1})
ksg = loss constant due to degradation (abiotic and biotic) (yr^{-1})
ksv = loss constant due to volatilization (yr^{-1})

Losses due to degradation (ksg) are empirically determined from field studies and should be available in literature. Soil losses due to leaching (ksl) and volatilization (ksv) can be calculated as shown below.

The soil loss constant of contaminants due to leaching (yr^{-1}) is obtained by calculating the water availability in the soil and the influence of soil properties on water and contaminant movement by the following equation:

$$ksl = \frac{P + I - Ev}{\theta \cdot Z \cdot [1.0 + (BD \cdot K_d/\theta)]} \qquad \text{(Equation 4-3)}$$

where:

ksl = loss constant due to leaching (yr^{-1})
P = average annual precipitation (cm/yr)
I = average annual irrigation (cm/yr)
Ev = average annual evapotranspiration (cm/yr)
θ = soil volumetric water content (mL/cm^3)
Z = soil depth from which leaching removal occurs (cm)
BD = soil bulk density (g/cm^3)
K_d = soil water partitioning coefficient (mL/g)

The soil loss constant due to volatilization (ksv) is based on gas equilibrium coefficients and gas phase mass transfer:

$$ksv = Ke \cdot Kt \qquad \text{(Equation 4-4)}$$

where:

ksv = soil loss constant due to volatilization (yr^{-1})
Ke = equilibrium coefficient
Kt = gas phase mass transfer coefficient (cm/s)

The equilibrium coefficient is calculated using the following equation:

$$Ke = \frac{3.1536x10^{10} \cdot H}{Z \cdot K_d \cdot R \cdot T \cdot BD} \qquad \text{(Equation 4-5)}$$

where:

Ke = equilibrium coefficient
H = Henry's law constant ($atm\text{-}m^3/mole$)
Z = soil depth (cm)
K_d = soil-water partitioning coefficient (mL/g)
R = ideal gas constant (L-atm/mole-°K)
T = temperature (°K)
BD = bulk density (g/cm^3)

The gas phase mass transfer can be calculated by the following equation:

$$Kt = 0.482 \cdot u^{0.78} \cdot N^{-0.67} \cdot d_e^{-0.11} \qquad \text{(Equation 4-6)}$$

where:

Kt = gas phase mass transfer coefficient (cm/s)
u = wind speed (m/s)
N = Schmidt number for gas phase (unitless)
d_e = effective diameter of contaminated area (m)

To calculate the Schmidt number for gas phase, the following equation can be used:

$$N = \frac{\mu_a}{pa \cdot D_a} \qquad \text{(Equation 4-7)}$$

where:

N = Schmidt number for gas phase (unitless)
μ_a = viscosity of air (g/cm-s)
pa = air density (g/cm^3)
D_a = diffusion coefficient of pollutant in air (cm^2/s)

The rate constant due to degradation (ksg) is based on field observations or empirical data and is discussed in more detail in section 4.3.5.2.

4.3. DESCRIPTION OF INPUT VARIABLES

Calculating cumulative soil concentration requires the input of site-specific data, which must be calculated or derived from the available literature. This section defines the input variables required for the calculations, discusses the factors that affect the value of each variable, and outlines the typical range of values and sources of each variable.

4.3.1. Dry and Wet Deposition Rates (Dyd and Dyw). Site-specific dry and wet deposition rates are determined by air dispersion modeling. Several methods for determining deposition rates are described in Chapter 3.

4.3.2. Time Period of Deposition (Tc). In this methodology, individuals are assumed to be exposed to contaminants in the soil following a time period in which there has been continuous combustor emissions. The lifetime of a combustion facility, such as a MWC, could be considered to be ≥ 30 years. However, since the site would already be dedicated to incineration, the combustor lifetime could be assumed to be as great as 100 years. In this document, 30, 60, and 100 years of continuous deposition will be used to construct Scenario A, Scenario B and Scenario C, respectively.

4.3.3. Soil Depth (Z). When modeling exposures to pollutants found in soils, the depth of assumed contamination is important in calculating the appropriate soil concentration. Contaminants deposited onto soil surfaces may be incorporated into lower soil profiles by tilling, whether done manually in a garden, or mechanically in a large field. In general, if the area under consideration is likely to be tilled (e.g., agricultural soils), soil depth may be assumed to be 10 - 20 cm, depending on local conditions and the equipment used. This methodology uses an incorporation depth of 20 cm for tilled soils. If soil incorporation does not occur, contaminants are assumed to be retained in a shallower, uppermost soil layer. While the actual depth of this uppermost layer is unknown, a value of 1 cm will be assumed in this methodology. If more specific information is available on soil mixing depth for a particular location, a different value can be substituted.

Soil concentrations based on soil depth are used to calculate exposure via several pathways: ingestion of plants contaminated by root uptake and by volatilization from the soil, direct ingestion of the soil by humans or herbivores, surface runoff into water bodies and dermal contact with the soil. The assumed soil depth varies according to the pathway under consideration. When calculating exposures due to uptake through plant roots, the average concentration of pollutant over the depth of the plant root determines plant uptake. For agricultural soils, the root depth is assumed to equal the tilling depth. For untilled soils (e.g., grazing pastures) the root zone does not directly reflect a tilling depth. However, this methodology assumes that tilling depth is an adequate substitute for root zone depth. If site-specific information is available for root zone depth, it can be used as the depth of contamination.

Plant uptake due to volatilization is assumed to occur from the uppermost soil layer. Any volatile contaminants are likely to be desorbed from particles soon after emission from the combustor, prior to deposition onto the soil. However, semivolatile compounds and volatiles emitted in sufficiently high concentrations may be deposited in particulate form and exhibit volatilization losses from soils. Pollutants subject to volatilization losses may be incorporated to 20 cm by tilling and will not readily volatilize from this depth. The volatilization rate will reflect the contaminant concentration at the soil surface.

For direct ingestion of soil by humans and herbivores, a depth of 1 cm is used. Children who either intentionally or incidentally eat soil are most likely to be exposed to contaminated soil in gardens, lawns, landscaped areas, parks and recreational areas. Herbivorous animals are most likely to be exposed to contaminated soils while grazing pastures. Most areas where soil exposure could occur are untilled or are tilled irregularly (e.g., pastures, lawns, parks), so the appropriate depth assumption ordinarily is 1 cm.

For dermal exposure, the conditions and areas of likely exposure are similar to those of the soil ingestion pathway. Therefore, as in the soil ingestion pathway, the dermal exposure model usually assumes a depth of 1 cm in estimating risk due to contact with contaminated soil. This assumption overestimates soil concentrations where incorporation occurs to any depth greater than 1 cm, and need not be used if site-specific information is available.

For exposure through the surface water, contaminants are assumed to travel from a watershed into a surface water body dissolved in runoff water and adsorbed to eroded soil particles. The assumptions for soil depth for the watershed area are site-specific. If the area under consideration is an agricultural area that is likely to be tilled, soil depth is assumed to be 20 cm. In all other cases, soil depth is assumed to be 1 cm, to be conservative.

4.3.4. Bulk Density of the Soil (BD). The soil bulk density (BD) is the ratio of the mass of the soil to its total volume. Bulk density is affected by the soil structure (e.g., the looseness or compaction of the soil, depending on the water and clay content of the soil) (Hillel, 1980). The values range from 0.93 - 1.84 g/cm^3 depending on soil type (Hoffman and Baes, 1979). This document assumes a bulk density of 1.5 g/cm^3. However, bulk density of the soil at the study site should be used, if available.

4.3.5. Soil Loss Constant (ks). Organic and some inorganic contaminants may be lost from the soil by many processes, which may occur simultaneously and/or at different rates. The rate at which a contaminant is lost from the soil is known as the "soil loss constant" and is determined by the physical, chemical and biological characteristics of the soil. The processes that remove contaminants from the soil include leaching, biotic and abiotic degradation, and volatilization.

Organics are subject to loss by leaching, degradation and volatilization. Inorganics, such as metals, may be lost by leaching and in some cases, by volatilization. This section discusses the kinetics of contamination loss, including first-order processes, which this methodology generally assumes, and other types of kinetics that may govern loss of contaminants from soil.

First-order reaction rates depend on the concentration of one reactant (Bohn et al., 1985). The loss of a contaminant by a first-order process depends only on the concentration of the contaminant in the soil, and a constant fraction of the compound is removed from the soil over time.

Those processes that apparently exhibit first-order kinetics without implying a mechanistic dependence on a first-order loss rate are termed "apparent first-order" loss rates (Sparks, 1989). Some higher-order kinetic processes appear to be first-order because of the relative abundance of some reactants, such as water and oxygen, in comparison with one reactant. This type of reaction is limited by the concentration of only one reactant under the conditions at which the apparent rate law is applicable (Sparks, 1989).

The assumption that contaminant loss follows first-order kinetics may be an oversimplification since at various concentrations or environmental conditions, the loss rates from soil systems will resemble different kinetic expressions. However, at low concentrations, a first-order loss constant may be adequate to describe the loss of the chemical from the soil.

In addition to first-order kinetics, contaminant loss in soil could follow zero-order or second-order kinetics. Zero-order loss rates are independent of reactant concentrations (Bohn et al., 1985). Zero-order rates describe some processes in which the reactants are in very high concentration. Under zero-order kinetics, a constant amount of a contaminant is lost from the soil over time, independent of its concentration.

Processes that follow second-order kinetics are dependent on the concentrations of two reactants or on the concentration of one reactant squared (Bohn et al., 1985). The loss constant of contaminant disappearance following a second-order process can be contingent on its own concentration, or on both its concentration and the concentration of another reactant, such as an enzyme or catalyst.

Because contaminant loss from soil is dependent on many complex factors, it may be difficult to model the overall rate of loss. Combined soil loss rates by these processes can be derived experimentally and values for certain compounds are presented in U.S. EPA (1986d). If field data where soil concentrations have been followed over time are available, these data should be used to calculate the soil concentration. Loss of the contaminant can be measured as the total loss by the three processes. In this case it would not be necessary to calculate the constant as described in Section 4.2.2. If the loss constant must be calculated, the input variables described below should be used. It should be noted that physical phenomena that may cause losses of contaminants from soils, such as leaching, volatilization and degradation can simultaneously occur, causing an overestimation of loss rates for each process (Valentine, 1986). When possible, the common occurrence of all loss processes should be taken into account.

4.3.5.1. SOIL LOSS DUE TO LEACHING (ksl) -- Losses of soil contaminants due to leaching are dependent on the amount of water available to generate leachate and on soil properties such as bulk density, soil porosity, and sorptive properties. Equation 4-3 in section 4.2.2. shows the calculation of ksl.

4.3.5.1.1. Available Water (P + I - Ev) -- The average annual volume of water available to generate leachate is the sum of the average annual precipitation (P) and the average annual irrigation (I) minus the average annual evapotranspiration (Ev). Precipitation values range from 18.06 to 164.19 cm/year (data for 69 selected cities; U.S. Bureau of the Census, 1987). Irrigation values range from 0 to 100 cm/year (data for 69 selected cities; Baes et al., 1984) Evapotranspiration values range from 35 to 100 cm/year (Baes et al., 1984). The available water as calculated by (P + I - Ev) is assumed to be a mass balance of all water inputs and outputs from the area in consideration. Other mechanisms of removal or retention in the soil are not considered in this methodology.

4.3.5.1.2. Soil Properties -- Three variables take into account the soil properties that affect leachate formation: bulk density (BD), soil water content (θ), and the soil-water partitioning coefficient (K_d). Bulk density is described in Section 4.3.4. The soil water content (θ) depends on the available water and on the soil structure. Values for θ range from 0.03 to 0.40 mL/cm^3 depending on soil type (Hoffman and Baes, 1979). The equilibrium partitioning coefficient (K_d), used to calculate the infiltration rate, reflects the sorption/desorption kinetics of the soil. Mean values for K_d for many chemicals can be found in Baes et al. (1984). However, variations in soil type greatly effect K_d, which can vary up to three orders of magnitude in soils ranging from pH 4.5 - 9.0. Therefore, it is advisable to use site-specific data for K_d when available. Partitioning coefficients for inorganic pollutants used in modeling may be derived from soil-to-plant transfer coefficients (Baes et al., 1984). Partitioning coefficients for organics may be estimated from K_{oc}, the K_d correlated to the soil organic carbon content, if the contaminant concentrations are in the appropriate range (<1/2 solubility) (Karickhoff, 1981).

4.3.5.2. SOIL LOSS DUE TO DEGRADATION (ksg) AND VOLATILIZATION (ksv) -- These variables reflect contaminant loss from the soil by processes other than leaching. These loss parameters are chemical-specific and are not applicable to most metals, which are transformed but not degraded by such mechanisms.

Equation 4-4 in section 4.2.2. shows the calculation of ksv. The soil loss due to degradation is based on field observations or empirical data as discussed below.

4.3.5.2.1. Loss Due to Biotic Degradation -- The rate of biological degradation in soils depends upon the concentration and activity of the microbial populations in the soil, the soil conditions, and the contaminant concentration (Jury and Valentine, 1986). First-order loss rates often do not account for the high variability of these parameters in a single soil system. However, the use of simple rate expressions may be appropriate at low chemical concentrations (ng range) where a first-order dependence on chemical concentration may be reasonable.

The rate of biological degradation of contaminants is chemical specific, depending on the complexity of the chemical and the usefulness of the chemical to the microorganisms. Some substrates are co-metabolized, not used by the organisms as a nutrient or energy source, but simply degraded along with other similar chemicals, which can be further utilized. Environmental and chemical factors that may limit the biodegradation of chemicals in the soil environment include availability of the chemical, nutrient limitations, toxicity of the compound, and inactivation or nonexistence of the enzyme capable of degrading the compound (Valentine and Schnoor, 1986).

4.3.5.2.2. Losses by Abiotic Degradation -- Chemical degradation of organic pollutants can be a significant mechanism of removal of these compounds from soil systems. Hydrolysis and oxidation-reduction (redox) reactions are the primary chemical transformation processes occurring in the upper layers of soils (Valentine, 1986). General rate expressions describing transformation of some contaminants by all nonbiological processes are available, and are helpful when division into component reactions is not possible.

Hydrolysis in aqueous systems is characterized by three processes: acid-catalyzed, base-catalyzed and neutral reactions (i.e., H^+, OH^- and H_2O catalyzed hydrolysis). The overall rate of hydrolysis is the sum of the first-order rates of these processes (Valentine, 1986). In soil systems, sorption of the chemical can increase, decrease or not affect its rate of hydrolysis as numerous studies cited in Valentine (1986) have shown. The total rate of hydrolysis in soil can be predicted by adding the rates in the soil and water phases, which are assumed first-order at fixed pH (Valentine, 1986). Methods for estimating these hydrolysis constants are described by Lyman et al. (1982).

Organic and inorganic compounds also undergo redox reactions in the soil (Valentine, 1986). Organic redox reactions involve exchange of oxygen and hydrogen atoms by the reacting molecules. For example, glucose is oxidized to CO_2 and H_2O by oxygen; and the oxygen is reduced to H_2O. Inorganic redox reactions may involve the exchange of atoms or of electrons by the reactants.

In soil systems where the identities of oxidant and reductant species are not specified, a first-order rate constant can be obtained for describing loss by redox reactions (Valentine, 1986). Redox reactions involving metals do not constitute a degradative process, but may promote losses by making metals more mobile. However, the effect of each redox reaction depends on the metal and on the soil conditions.

4.3.5.2.4. Losses Due to Volatilization (ksv) -- Semivolatile and volatile compounds emitted in high concentrations may become adsorbed to soil particles and exhibit volatilization losses from the soil. The loss of a chemical from the soil by volatilization is dependent on the rate of movement of the chemical to the soil surface, the chemical vapor concentration at the soil surface (due to soil conditions) and the rate at which the vapor is carried away by the atmosphere (due to atmospheric conditions) (Jury, 1986).

Polar molecules are less volatile than slightly polar or nonpolar molecules due to greater absorption to soil constituents and greater water solubility (Poe, 1988). Volatilization loss rates for specific chemicals may be found in the literature. Volatilization rates for hexachlorobenzene have been modeled by Spencer et al. (1982). Laboratory measured volatilization rates for soil incorporated chemicals adequately estimate the field measured rates, because the rates are primarily dependent on soil factors that can be mimicked in the laboratory (Spencer et al., 1982). Such experimental values may be used if site-specific data are unavailable.

4.4. EXAMPLE CALCULATIONS

In determining soil concentration, the time of deposition (i.e., the number of years of continual combustor emissions) and the distance of the individual from the incinerator differ for each exposure scenario. In Scenario A the duration of deposition is 30 years, the annual deposition value is calculated as the areal average over the 50 km "ring" surrounding the incinerator, and the exposed individual is within this 50 km radius. In Scenario B, the individual is 5 km from the combustion source exposed to the areal-averaged annual deposition for the 5 km ring and the duration of deposition is 60 years. Scenario C defines the individual as residing at the point of maximal deposition at a distance of 0.2 km from the incinerator and the duration of deposition as 100 years. This section illustrates the soil concentration for the three exposure scenarios (A,B,C) for both cadmium and benzo(a)pyrene (B(a)P).

4.4.1. Cadmium. The values chosen for input variables to determine soil concentration of cadmium are found in Table 4-2.

In determining soil concentrations, the soil loss constant of cadmium due to leaching is first calculated. For surface depth (Z = 1 cm), the soil loss constant (ksl) can be calculated according to Equation 4-3:

$$\mathrm{ksl} = \frac{120.0\ \mathrm{cm/yr} + 25.0\ \mathrm{cm/yr} - 100.0\ \mathrm{cm/yr}}{0.22\ \mathrm{mL/cm^3} \cdot 1\ \mathrm{cm} \cdot [1.0 + (1.5\ \mathrm{g/cm^3} \cdot 500\ \mathrm{mL/g}/0.22\ \mathrm{mL/cm^3})]}$$

$$= 0.06\ \mathrm{yr^{-1}}$$

The soil loss constant, due to all processes, for cadmium at surface depth (Z = 1 cm) can be determined by using Equation 4-2:

$$\mathrm{ks} = 0.06\ \mathrm{yr^{-1}} + 0\ \mathrm{yr^{-1}} + 0\ \mathrm{yr^{-1}}$$

$$= 0.06\ \mathrm{yr^{-1}}$$

TABLE 4-2

Input Variables for Determination of Cadmium Soil Concentrations

Variable	Exposure Scenario		
	A	B	C
P (cm/yr)	120.0	120.0	120.0
I (cm/yr)	25.0	25.0	25.0
Ev (cm/yr)	100.0	100.0	100.0
θ (mL/cm^3)	0.22	0.22	0.22
Z (cm)			
Surface depth	1	1	1
Till depth	20	20	20
BD (g/cm^3)	1.5	1.5	1.5
K_d (mL/g)	500	500	500
ksg (yr^{-1})	0	0	0
ksv (yr^{-1})	0	0	0
Dyd (g/m^2/yr)	$2.34\text{x}10^{-5}$	$1.59\text{x}10^{-4}$	$0.09\text{x}10^{-2}$
Dyw (g/m^2/yr)	$0.66\text{x}10^{-5}$	$2.17\text{x}10^{-4}$	$1.10\text{x}10^{-2}$
Tc (yrs)	30	60	100

The soil loss constant for cadmium due to leaching at till depth (Z = 20 cm) can be determined using Equation 4-3:

$$ksl = \frac{120.0\ \text{cm/yr} + 25.0\ \text{cm/yr} - 100.0\ \text{cm/yr}}{0.22\ \text{mL/cm}^3 \cdot 20\ \text{cm} \cdot [1.0 + (1.5\ \text{g/cm}^3 \cdot 500\ \text{mL/g}/0.22\ \text{mL/cm}^3)]}$$

$$= 0.003\ \text{yr}^{-1}$$

The soil loss constant, due to all processes, for cadmium at till depth (Z =20 cm) can be calculated according to Equation 4-2:

$$ks = 0.003\ \text{yr}^{-1} + 0\ \text{yr}^{-1} + 0\ \text{yr}^{-1}$$

$$= 0.003\ \text{yr}^{-1}$$

In calculating the soil concentration in Scenario A, it is assumed that the incinerator has been operating continuously for 30 years (Tc = 30 yrs). The soil concentrations at the surface and till depths can be determined by using Equation 4-1. For surface depth (Z = 1 cm), soil concentration is:

$$Sc = \frac{(2.34\text{x}10^{-5} + 0.66\text{x}10^{-5}) \cdot [1.0-\exp(-0.06\ \text{yr}^{-1} \cdot 30\ \text{yrs})] \cdot 0.1}{1\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.06\ \text{yr}^{-1}}$$

$$= 2.78\text{x}10^{-5}\ \text{mg Cd/g soil}$$

Soil concentration for till depth (Z = 20 cm) is:

$$Sc = \frac{(2.34\text{x}10^{-5} + 0.66\text{x}10^{-5}) \cdot [1.0\text{-exp}(-0.003\ \text{yr}^{-1} \cdot 30\ \text{yrs})] \cdot 0.1}{20\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.003\ \text{yr}^{-1}}$$

$= 2.87\text{x}10^{-6}$ mg Cd/g soil

In Scenario B, it is assumed that the incinerator has been operating continuously for 60 years. The soil concentration for both surface depth and till depth can be calculated using Equation 4-1. For surface depth (Z= 1 cm) soil concentration is:

$$Sc = \frac{(1.59\text{x}10^{-4} + 2.17\text{x}10^{-4}) \cdot [1.0\text{-exp}(-0.06\ \text{yr}^{-1} \cdot 60\ \text{yrs})] \cdot 0.1}{1\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.06\ \text{yr}^{-1}}$$

$= 4.06\text{x}10^{-4}$ mg Cd/g soil

For till depth (Z = 20 cm), soil concentration is:

$$Sc = \frac{(1.59\text{x}10^{-4} + 2.17\text{x}10^{-4}) \cdot [1.0\text{-exp}(-0.003\ \text{yr}^{-1} \cdot 60\ \text{yrs})] \cdot 0.1}{20\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.003\ \text{yrs}^{-1}}$$

$= 6.88\text{x}10^{-5}$ mg Cd/g soil

In Scenario C, the combustion source has been operating for 100 years. The soil concentration of cadmium in Scenario C for surface and till depth can be calculated using Equation 4-1. For surface depth (Z = 1 cm), soil concentration is:

$$Sc = \frac{(0.09\text{x}10^{-2} + 1.10\text{x}10^{-2}) \cdot [1.0 - \exp(-0.06\ \text{yr}^{-1} \cdot 100\ \text{yrs})] \cdot 0.1}{1\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.06\ \text{yr}^{-1}}$$

$$= 1.32\text{x}10^{-2}\ \text{mg Cd/g soil}$$

For till depth (Z = 20 cm), soil concentration is:

$$Sc = \frac{(0.09\text{x}10^{-2} + 1.10\text{x}10^{-2}) \cdot [1.0 - \exp(-0.003\ \text{yr}^{-1} \cdot 100\ \text{yrs})] \cdot 0.1}{20\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.003\ \text{yr}^{-1}}$$

$$= 3.43\text{x}10^{-3}\ \text{mg Cd/g soil}$$

4.4.2. Benzo(a)pyrene (B(a)P). The values chosen for input variables to determine soil concentration of benzo(a)pyrene are found in Table 4-3.

The soil loss constant of benzo(a)pyrene due to leaching for surface depth (Z = 1 cm) can be calculated according to Equation 4-3:

$$ksl = \frac{120.0\ \text{cm/yr} + 25.0\ \text{cm/yr} - 100.0\ \text{cm/yr}}{0.22\ \text{mL/cm}^3 \cdot 1\ \text{cm} \cdot [1.0 + (1.5\ \text{g/cm}^3 \cdot 1.2\text{x}10^5\ \text{mL/g}/0.22\ \text{mL/cm}^3)]}$$

$$= 2.5\text{x}10^{-4}\ \text{yr}^{-1}$$

The soil loss constant of benzo(a)pyrene due to leaching at till depth (Z = 20 cm) can be determined using Equation 4-3:

$$ksl = \frac{120.0\ \text{cm/yr} + 25.0\ \text{cm/yr} - 100.0\ \text{cm/yr}}{0.22\ \text{mL/cm}^3 \cdot 20\ \text{cm} \cdot [1.0 + (1.5\ \text{g/cm}^3 \cdot 1.2\text{x}10^5\ \text{mL/g}/0.22\ \text{mL/cm}^3)]}$$

$$= 1.25\text{x}10^{-5}\ \text{yr}^{-1}$$

TABLE 4-3

Input Variables for Determination of Benzo(a)Pyrene Soil Concentrations

	Exposure Scenario		
Variable	A	B	C
P (cm/yr)	120.0	120.0	120.0
I (cm/yr)	25.0	25.0	25.0
Ev (cm/yr)	100.0	100.0	100.0
θ (mL/cm^3)	0.22	0.22	0.22
Z (cm)			
Surface depth	1	1	1
Till depth	20	20	20
BD (g/cm^3)	1.5	1.5	1.5
K_d (mL/g)	1.2x10^5	1.2x10^5	1.2x10^5
ksg (yr^{-1})	0.803	0.803	0.803
Dyd (g/m^2/yr)	3.77x10^{-8}	2.45x10^{-7}	0.142x10^{-5}
Dyw (g/m^2/yr)	2.74x10^{-8}	6.74x10^{-7}	2.33x10^{-5}
Tc (yrs)	30	60	100
μ_a (g/cm-s)	184	184	184
pa (g/cm^3)	1.19x10^{-3}	1.19x10^{-3}	1.19x10^{-3}
D_a (cm^2/s)	0.043	0.043	0.043
u (m/s)	3.9	3.9	3.9
d_e (m)	200	200	200
H (atm-m^3/mole)	1.55x10^{-6}	1.55x10^{-6}	1.55x10^{-6}
R (L-atm/mole-°K)	0.08205	0.08205	0.08205
T (°K)	298	298	298

The loss constant of benzo(a)pyrene due to volatilization (ksv) requires that two variables be calculated, the gas phase mass transfer coefficient and the equilibrium coefficient. The Schmidt number (N), needed as an input for the equilibrium coefficient, can be calculated using Equation 4-7:

$$N = \frac{184 \text{ g/cm-sec}}{1.19\text{x}10^{-3} \text{ g/cm}^3 \cdot 0.043 \text{ cm}^2\text{/s}}$$

$$= 3.59\text{x}10^6$$

The gas phase mass transfer (Kt) can be determined using Equation 4-6:

$$Kt = 0.482 \cdot (3.9 \text{ m/s})^{0.78} \cdot (3.59\text{x}10^6)^{-0.67} \cdot (200 \text{ m})^{-0.11}$$
$$= 3.16\text{x}10^{-5} \text{ cm/s}$$

The equilibrium coefficient (Ke) for surface depth (Z = 1 cm) can be determined using Equation 4-5:

$$Ke = \frac{3.1536\text{x}10^{10} \cdot 1.55\text{x}10^{-6} \text{ (atm-m}^3\text{/mole)}}{1 \text{ cm} \cdot 1.2\text{x}10^5 \text{ mL/g} \cdot 0.08205 \text{ (L-atm/mole-°K)} \cdot 298\text{°K} \cdot 1.5 \text{ g/cm}^3}$$
$$= 1.11\text{x}10^{-2}$$

The equilibrium coefficient (Ke) at till depth (Z = 20 cm) can be calculated using Equation 4-5:

$$Ke = \frac{3.1536\text{x}10^{10} \cdot 1.55\text{x}10^{-6} \text{ (atm-m}^3\text{/mole)}}{20 \text{ cm} \cdot 1.2\text{x}10^5 \text{ mL/g} \cdot 0.08205 \text{ (L-atm/mole-°K)} \cdot 298\text{°K} \cdot 1.5 \text{ g/cm}^3}$$

$$= 5.5\text{x}10^{-4}$$

The soil loss of benzo(a)pyrene due to volatilization (ksv) at surface depth (Z = 1 cm) can be calculated using Equation 4-4:

$$\begin{aligned} ksv &= 1.11\text{x}10^{-2} \cdot 3.16\text{x}10^{-5} \\ &= 3.51\text{x}10^{-7}\ yr^{-1} \end{aligned}$$

The soil loss of benzo(a)pyrene due to volatilization (ksv) at till depth (Z = 20 cm) can be calculated using Equation 4-4:

$$\begin{aligned} ksv &= 5.5\text{x}10^{-4} \cdot 3.16\text{x}10^{-5} \\ &= 1.74\text{x}10^{-8}\ yr^{-1} \end{aligned}$$

The soil loss constant of benzo(a)pyrene due to degradation (ksg) is equal to 0.803 yr^{-1}, which is experimentally derived based on the half-life of benzo(a)pyrene in soil (U.S. EPA, 1988c). The amount of benzo(a)pyrene lost to volatilization and leaching is minimal relative to the amount lost due to degradation.

The soil loss constant of benzo(a)pyrene due to all processes (ks) at surface depth (Z = 1 cm) can be calculated using Equation 4-2:

$$\begin{aligned} ks &= 2.5\text{x}10^{-4}\ yr^{-1} + 0.803\ yr^{-1} + 3.51\text{x}10^{-7}\ yr^{-1} \\ &= 0.803\ yr^{-1} \end{aligned}$$

The soil loss constant of benzo(a)pyrene due to all processes (ks) at till depth (Z = 20 cm) can be calculated using Equation 4-2:

$$\begin{aligned} ks &= 1.25\text{x}10^{-5}\ yr^{-1} + 0.803\ yr^{-1} + 1.74\text{x}10^{-8}\ yr^{-1} \\ &= 0.803\ yr^{-1} \end{aligned}$$

In calculating the soil concentration in Scenario A, it is assumed that the incinerator has been operating continuously for 30 years (Tc = 30 yrs). The soil concentrations at the surface and till depths can be determined by using Equation 4-1. For surface depth (Z = 1 cm), soil concentration is:

$$Sc = \frac{(3.77\text{x}10^{-8} + 2.74\text{x}10^{-8}) \cdot [1.0\text{-exp}(-0.803\ \text{yr}^{-1} \cdot 30\ \text{yrs})] \cdot 0.1}{1\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.803\ \text{yr}^{-1}}$$

$= 5.43\text{x}10^{-9}$ mg B(a)P/g soil

For till depth (Z = 20 cm), soil concentration is:

$$Sc = \frac{(3.77\text{x}10^{-8} + 2.74\text{x}10^{-8}) \cdot [1.0\text{-exp}(-0.803\ \text{yr}^{-1} \cdot 30\ \text{yrs})] \cdot 0.1}{20\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.803\ \text{yr}^{-1}}$$

$= 2.70\text{x}10^{-10}$ mg B(a)P/g soil

In Scenario B, it is assumed that the incinerator has been operating continuously for 60 years. The soil concentration for both surface depth and till depth can be calculated using Equation 4-1. For surface depth (Z= 1 cm), soil concentration is:

$$Sc = \frac{(2.45\text{x}10^{-7} + 6.74\text{x}10^{-7}) \cdot [1.0\text{-exp}(-0.803\ \text{yr}^{-1} \cdot 60\ \text{yrs})] \cdot 0.1}{1\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.803\ \text{yr}^{-1}}$$

$= 7.66\text{x}10^{-8}$ mg B(a)P/g soil

For till depth (Z = 20 cm), soil concentration is:

$$Sc = \frac{(2.45\text{x}10^{-7} + 6.74\text{x}10^{-7}) \cdot [1.0\text{-exp}(-0.803\ \text{yr}^{-1} \cdot 60\ \text{yrs})] \cdot 0.1}{20\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.803\ \text{yrs}^{-1}}$$

$= 3.81\text{x}10^{-9}$ mg B(a)P/g soil

In Scenario C, the combustion source has been operating for 100 years. The soil concentration of cadmium in Scenario C for surface and till depth can be calculated using Equation 4-1. For surface depth (Z = 1 cm), soil concentration is:

$$Sc = \frac{(0.142\text{x}10^{-5} + 2.33\text{x}10^{-5}) \cdot [1.0\text{-exp}(-0.803\ \text{yr}^{-1} \cdot 100\ \text{yrs})] \cdot 0.1}{1\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.803\ \text{yr}^{-1}}$$

$= 2.06\text{x}10^{-6}$ mg B(a)P/g soil

For till depth (Z = 20 cm), soil concentration is:

$$Sc = \frac{(0.142\text{x}10^{-5} + 2.33\text{x}10^{-5}) \cdot [1.0\text{-exp}(-0.803\ \text{yr}^{-1} \cdot 100\ \text{yrs})] \cdot 0.1}{20\ \text{cm} \cdot 1.5\ \text{g/cm}^3 \cdot 0.803\ \text{yr}^{-1}}$$

$= 1.02\text{x}10^{-7}$ mg B(a)P/g soil

The soil concentrations determined above and shown in Table 4-4 are used in subsequent chapters as input variables for determination of exposure by various pathways.

TABLE 4-4

Summary of Soil Concentrations
(mg/g)

	Cadmium	Benzo(a)pyrene
Scenario A:		
1 cm depth	2.78×10^{-5}	5.43×10^{-9}
20 cm depth	2.87×10^{-6}	2.70×10^{-10}
Scenario B:		
1 cm depth	4.06×10^{-4}	7.66×10^{-8}
20 cm depth	6.88×10^{-5}	3.81×10^{-9}
Scenario C:		
1 cm depth	1.32×10^{-2}	2.06×10^{-6}
20 cm depth	3.43×10^{-3}	1.02×10^{-7}

5. DETERMINING EXPOSURE THROUGH THE TERRESTRIAL FOOD CHAIN

5.1. INTRODUCTION

The previous chapter describes how contaminants associated with combustor emissions are incorporated into soil. This chapter describes how contaminants may enter the terrestrial food chain (TFC) from the soil or directly from the atmosphere. It also shows how daily exposure to food chain contaminants is calculated and used to estimate human risk. The model assumes that contaminants enter the food chain through either uptake by plants or direct soil ingestion by animals. Humans may also directly ingest soil; however, this pathway of exposure is discussed in Chapter 6. The remaining sections of this chapter discuss methods and important parameters for calculating pollutant concentrations in plant and animal tissues and determining human daily intake from foods. Example calculations for benzo(a)pyrene and cadmium are also presented.

5.2. CALCULATING CONCENTRATION OF POLLUTANT IN PLANTS

This chapter examines three mechanisms by which pollutants can bioaccumulate in plants: uptake by roots, direct deposition on exposed plant tissues, and air-to-plant transfer of vapor-phase pollutants. The relative importance of each mechanism depends in part on which of three broad categories a plant belongs to: leafy vegetables, exposed produce, or protected produce. Available research on mechanisms of pollutant uptake by the three routes is limited. Questions exist as to whether pollutants accumulated by direct deposition and air-to-plant transfer remain on the plant surface or are internalized by plants. In addition, the degree to which pollutants are transported between plant organs through the vascular system is uncertain. Thus, uptake via direct deposition and air-to-plant transfer is calculated for leafy and exposed produce but not protected produce since the latter is not in direct contact with the air.

Yang and Nelson (1986) describe leafy vegetables, such as lettuce, as having a "broad flat leaf for the direct interception of atmospherically deposited material." Exposed produce, such as fruits, has a smaller exposed surface area per unit mass than leafy produce although it still intercepts atmospherically deposited particles. Protected produce, such as potatoes or vegetables in pods, is not directly exposed to atmospherically deposited material. For this plant category, accumulation is most likely to occur by root uptake.

The food chain model calculates the pollutant concentration in each plant due to the appropriate mechanism and sums them to obtain the total pollutant concentration:

$$P_i = Pr_i + Pd_i + Pv_i \qquad \text{(Equation 5-1)}$$

where:

P_i = total concentration of pollutant in the i^{th} plant group ($\mu g/g$)
Pr_i = concentration of pollutant in i^{th} plant group due to root uptake ($\mu g/g$)
Pd_i = concentration of pollutant in i^{th} plant group due to direct deposition ($\mu g/g$)
Pv_i = concentration of pollutant in i^{th} plant group due to air-to-plant transfer ($\mu g/g$)

Section 5.5 presents calculations of P_i for seven plant groups consumed by humans (grains, legumes, potatoes, root vegetables, fruiting vegetables, leafy vegetables, fruits) and two plant groups consumed by animals (grains and forage). Grains, potatoes, and root vegetables are considered to be protected produce; fruits, fruiting vegetables, and legumes are considered to be exposed produce; and leafy vegetables and forage are considered leafy produce. Risk assessors may want to consider site-specific data when developing appropriate plant groups. Note that a separate P_i must be calculated for each pollutant in each plant group to be considered in the risk assessment.

5.2.1. Plant Pollutant Concentration Due to Root Uptake (Pr_i).

The concentration of pollutant in plant tissue due to root uptake is determined from the soil concentration and the plant-soil bioconcentration factor for each plant group:

$$Pr_i = Sc \cdot Br_i \qquad \text{(Equation 5-2)}$$

where:

Pr_i = concentration of pollutant in i^{th} plant group due to root uptake (μg pollutant/g plant tissue, dry weight [DW])

Sc = soil concentration of pollutant after the total period of deposition (μg pollutant/g soil)

Br_i = plant-soil bioconcentration factor for the i^{th} plant group ([μg pollutant/g plant tissue DW]/[μg pollutant/g soil])

5.2.1.1. SOIL CONCENTRATION (Sc) -- Methods for calculating soil concentration are presented in Chapter 4. When considering root uptake, soil concentration should be calculated assuming an incorporation depth of 20 cm since the ground is usually tilled to this depth to plant food and grain crops. Even for crops such as pasture grasses grown on untilled soil where pollutants are assumed to be within the upper 1 cm of soil, plant roots penetrate well beyond this depth into relatively uncontaminated soil. Therefore, a soil depth of 20 cm is also used in these cases to estimate the average soil concentration in a 20-cm root zone.

5.2.1.2. BIOCONCENTRATION FACTORS AND PLANT UPTAKE (Br_i) -- The plant-soil bioconcentration factor is a measure of a pollutant's ability to accumulate in plant tissue and is defined as the pollutant concentration in the plant divided by the pollutant concentration in the soil. Bioconcentration factors may be derived differently for inorganic and organic pollutants, but they are generally dependent on the bioavailability of the pollutant in the soil.

5.2.1.2.1. Inorganics -- Data on uptake of inorganics by plants may be available in the form of bioconcentration factors, which are discussed in Baes et al. (1984). However, in some cases, accumulation of inorganics is measured by "uptake response slopes" rather than by bioconcentration factors.

The term "uptake response" is used to denote the difference between pre- and post-exposure plant tissue concentrations. Uptake response slopes can be derived from any data set where changes in tissue concentration and pollutant application rate have been determined; the units are often expressed as (μg/g

DW)/(kg/ha). Uptake response slopes can be converted to bioconcentration factors by multiplying by a conversion factor of 2.7×10^3 Mg/ha, which is the dry mass of soil in the upper 20 cm if a typical bulk density of 1.35 g/cm^3 is assumed.

Uptake rates of metals by plants, reviewed by CAST (1980), Ryan et al. (1982), and Logan and Chaney (1983), are influenced by application rate, soil pH and soil organic matter content. Ryan et al. (1982) used linear regression of plant tissue cadmium concentration against cadmium application rate to derive uptake response slopes for various crops. The authors stated that, given constant soil conditions, cadmium plant tissue concentration had a highly significant positive linear correlation with the amount of cadmium added to the soil.

More recent work, as reviewed by Page et al. (1986), has shown that plant response to metals from sludge-amended soil is curvilinear, approaching a plateau concentration in tissue as sludge application rate increases. However, metal-adsorptive materials present in the sludge matrix are thought to be responsible for this effect. Since no such uptake-limiting effect has been demonstrated for deposited metals, uptake response slopes will be assumed to be linear for this methodology. Therefore, plant concentration and resulting dietary intake of a contaminant are assumed to increase continually with deposition if some or all of the diet originates from contaminated soils. This assumption is conservative but reasonable given the lack of data regarding the saturability of plant tissues.

Soil pH is probably the most important characteristic influencing metal retention, and thus availability, by soils (LaBauve et al., 1988). The pH of soils varies greatly, from 2 or less in acid soils to 9 or more in alkaline soils (Hillel, 1980). Agricultural soils are usually limed to maintain the pH at approximately 6.5, the pH at which plant nutrients are most readily available for plant uptake (Bohn et al., 1985).

Metal cations exhibit a pH-dependent adsorption behavior, probably due to several different adsorption mechanisms (Elliott et al., 1986). At pH less than 6.0, some metal cations tend to be more mobile. For example, cadmium mobility tends to increase as soil pH decreases; lead is less pH-dependent (Harter, 1983).

As soil pH increases, generally above 6.0, metal cations are adsorbed more strongly and become less available for uptake by plants. Several studies have demonstrated that, generally, plant uptake of metals decreases with increasing pH (Lagerwerff, 1971; Ryan et al., 1982; Giordano et al., 1983). However, the effects of liming acid soils to increase pH are unclear. While Giordano et al. (1983) and Ryan et al. (1982) report decreased cadmium concentration in plants after soil has been limed to pH 6.5, several studies cited in CAST (1980) indicate that using lime to raise soil pH from 5.0 to 6.6 had little or no effect on metal uptake by plants.

Ideally, the risk assessor should use site-specific data because local soil pH and organic matter content greatly influence the uptake response. If literature data are used, the data should be from studies in which the pH and organic matter content of the experimental soils are similar to those of soils at the site. Where evidence suggests that response may be other than linear, a slope derived from application rates (or soil concentrations) similar to those predicted for the site in question should be preferred. Also, uptake response data should be derived from studies of deposited emissions, if possible. For contaminants lacking such data, uptake response slopes derived from other types of chemical application (such as sludge or pesticide additions) can be assumed to apply to deposited contaminants as well.

5.2.1.2.2. Organics -- A linear uptake response is also assumed for organic chemicals, but because most of the compounds of concern are xenobiotics, tissue concentration can be assumed to be zero when soil concentration is zero. Therefore, a bioconcentration factor, which can be derived from a single data pair, is used more often for organics than for inorganics. Such a factor can be derived from any data set that includes both tissue concentration (μg/g DW) and soil concentration (μg/g).

As for inorganics, however, the most accurate factor would be derived from interpolation of values for soil concentration similar to those predicted for the site of concern. Ideally, factors determined using local soil data should be used. However, if literature data are used, the experimental soils should be of similar type, pH, and organic matter as soils on the site to the greatest extent possible.

For organics, uptake from soil and transport to above-ground plant parts is dependent on the solubility of a chemical in water, which is inversely proportional to the octanol-water partition coefficient (K_{ow}) (Travis and Arms, 1988). Thus, the bioconcentration factor can be calculated from the following equation if literature data are unavailable:

$$\log Br_i = 1.588 - 0.578 \log K_{ow} \quad \text{(Equation 5-3)}$$

This equation is based on bioconcentration factors for 29 chemicals in vegetative tissues (Travis and Arms, 1988).

The bioconcentration factors for cadmium and benzo(a)pyrene used in the example calculations are presented in Belcher and Travis (1989). These factors are available for calculating uptake into potatoes, leafy vegetables, legumes, root vegetables, fruiting vegetables, grain and forage. The bioconcentration factor for fruiting vegetables was assumed to apply to fruits also because both plant groups belong to the same broad group of exposed produce. However, if site-specific data concerning uptake into fruits is available, this data should be used.

5.2.2. Plant Pollutant Concentration Due to Direct Deposition (Pd_i). Pollutants are deposited on exposed plant surfaces by both dry and wet deposition. Pollutant concentration due to deposition is calculated for leafy plants and exposed produce but not for protected produce (Yang and Nelson, 1986), which is not directly exposed to deposition. Pd contributes more to the total concentration of leafy produce than exposed produce because leafy produce has a larger surface area with which to intercept direct deposition (Yang and Nelson, 1986; Shor et al., 1982). Concentration of pollutant in plant tissue due to direct deposition onto plant surfaces is calculated as:

$$Pd_i = \frac{1000 \cdot [Dyd + (Fw \cdot Dyw)] \cdot Rp_i \cdot [1.0 - \exp(-kp \cdot Tp_i)]}{Yp_i \cdot kp}$$

(Equation 5-4)

where:

Pd_i = concentration of pollutant due to direct deposition in the i^{th} plant group (μg pollutant/g plant tissue DW)
1000 = conversion factor, kg/10^3 g and 10^6 μg/g pollutant
Dyd = yearly dry deposition rate (g pollutant/m^2/yr)
Fw = fraction of wet deposition that adheres to plant surfaces (unitless)
Dyw = yearly wet deposition rate (g pollutant/m^2/yr)
Rp_i = interception fraction of the edible portion of plant tissue for the i^{th} plant group (unitless)
kp = plant surface loss coefficient (yr^{-1})
Tp_i = length of plant's exposure to deposition per harvest of the edible portion of the i^{th} plant group (yrs)
Yp_i = yield or standing crop biomass of the edible portion of the i^{th} plant group (kg DW/m^2)

5.2.2.1. YEARLY DEPOSITION RATES (Dyd and Dyw) -- Refer to Chapter 3 to find methods for determining both dry and wet yearly deposition averaged over the area surrounding the combustor.

5.2.2.2. FRACTION OF WET DEPOSITION ADHERING (Fw) -- All of the pollutants that come in contact with plant surfaces as a result of dry deposition are assumed to remain on the plant surfaces until removed by weathering. However, only a fraction of pollutant material that intercepts plant surfaces as a result of wet deposition adheres; the remainder is assumed to wash off almost immediately. Fw is a measure of the amount of wet deposition that adheres to the plant surface. Fw is assumed to be a constant of 2% for all pollutants and plant groups (Belcher, 1989).

5.2.2.3. INTERCEPTION FRACTION (Rp_i) -- Baes et al. (1984) define the interception fraction as a "factor which accounts for the fact that not all of the airborne material depositing within a unit area will initially deposit on edible vegetation surfaces." Models developed for the Nuclear Regulatory Commission assumed the interception fraction was a constant of 0.2 for dry and wet deposition of particles (Boone et al., 1981). However, Shor et al. (1982) suggest that the diversity of plant growth indicates the need for vegetation-specific interception fractions. Baes et al. (1984) presents equations for interception fractions for pasture grasses, leafy vegetables, and exposed produce. Based on an empirically derived relationship to standing crop biomass, interception fractions for pasture grasses can be calculated as:

$$Rpg = 1-\exp(-2.88 \cdot Yp_i) \qquad \text{(Equation 5-5)}$$

where:

Rpg = interception fraction for pasture grasses (unitless)
Yp_i = standing crop biomass for pasture grasses (kg DW/m^2)

Interception fractions for mature leafy vegetables can be calculated as:

$$R_{mlv} = \frac{n_r \; r_n \; \pi r_f^2}{[(n_r-1)d_p + 2r_f][(r_n-1)d_r + 2r_f]}$$

(Equation 5-6)

where:

R_{mlv} = interception fraction for mature leafy vegetables (unitless)
n_r = number of plants per row
r_n = number of rows of plants
r_f = radius of individual fruit or plant (mm)
d_p = distance between plants in a row (mm)
d_r = distance between rows of plants (mm)

The average interception fraction for mature leafy vegetables was calculated to be 0.3. Interception fractions for exposed produce is based on the five most commercially important non-citrus fruit and field crops (apples, snap beans, tomatoes, peaches, cherries; Baes et al., 1984) and is calculated by determining the total fruit cross-sectional area per unit area:

$$R_{rf} = \frac{n\pi r_f^2}{lw} \qquad \text{(Equation 5-7)}$$

where:

R_{rf} = interception fraction of round fruit (unitless)
n = number of fruit per unit area
r_f = radius of the fruit (mm)
l = length of the unit area (mm)
w = width of the unit area (mm)

For long rather than round exposed produce (i.e., snap beans), the above equation is modified to:

$$R_{lf} = \frac{n2r_f l_f}{lw} \qquad \text{(Equation 5-8)}$$

where:

R_{lf} = interception fraction of long fruit (unitless)
l_f = length of the long fruit (mm)

Interception fractions for leafy vegetables and exposed produce can also be calculated based on an empirical relationship to standing crop biomass (Yp_i) as for pasture grasses:

$$R_{lv} = 1-\exp(-0.0846 \cdot Yp_i) \qquad \text{(Equation 5-9)}$$

$$R_e = 1-\exp(-0.0324 \cdot Yp_i) \qquad \text{(Equation 5-10)}$$

where:

R_{lv} = interception fraction for leafy vegetables (unitless)
R_e = interception fraction for exposed produce (unitless)

Risk assessors may wish to choose the appropriate set of equations for interception fraction based on the availability of site-specific data. In addition, values for number of plants per row, number of fruit per square meter, etc. should be selected from site-specific data.

5.2.2.4. PLANT-SURFACE LOSS COEFFICIENT (kp) -- Once contaminants are deposited onto plant surfaces, several environmental processes, including wind removal, water removal, and growth dilution, work to reduce the amount of contamination on the plant surface (Miller and Hoffman, 1983). The plant-surface loss coefficient is a measure of the amount of contaminant lost to these processes over time. Miller and Hoffman (1983) give the following equation to calculate kp:

$$kp = \frac{\ln 2}{t_{1/2}} \cdot 365 \text{ days/yr} \qquad \text{(Equation 5-11)}$$

where:

kp = plant-surface loss coefficient (yrs^{-1})
$t_{1/2}$ = environmental half-time (days)

In radiologic assessments, a half-time value of 14 days has generally been assumed to represent all radionuclides and plant types (Miller and Hoffman, 1983). However, Miller and Hoffman (1983) report half-time values ranging from 2.8 to 34 days for all contaminants on herbaceous vegetation. These half-time values correspond to kp values ranging from 7.44 to 90.36 yrs^{-1}. The half-time of benzo(a)pyrene in air is reported to range from 1-6 days (U.S. EPA, 1986d); values of kp calculated from those half lives range from 42-253 yr^{-1}. The value of 126.5 yr^{-1} was selected from this range to represent a typical value (Belcher and Travis, 1989).

5.2.2.5. LENGTH OF PLANT EXPOSURE (Tp) -- This variable is defined as the amount of time the edible part of the plant is exposed to direct deposition. Tp is treated as a constant and is based on an average period between successive hay harvests (60 days) and an average period between successive grazing by cattle (30 days) (Belcher and Travis, 1989). For animal forage, these two time periods are averaged (45 days) and then divided by 365 days to give a Tp of 0.123 yrs. For the remaining animal foods, time between successive hay harvests is divided by 365 days to give a Tp of 0.164 (Belcher and Travis, 1989). For lack of appropriate data, a Tp of 0.164 is also used in this document for foods consumed by humans. However, risk assessors may want to consider local growing seasons to develop site-specific Tp values for foods consumed by humans.

5.2.2.6. STANDING CROP BIOMASS (Yp_i) -- As may be seen from Equations 5-4 through 5-10, plant pollutant concentration due to direct deposition is influenced by the standing crop biomass, Yp_i. The best estimate of Yp_i is productivity (Baes et al., 1984; Shor et al., 1982), which is defined as:

$$Yp_i \approx P_i = \frac{Yh_i}{Ah_i} \qquad \text{(Equation 5-12)}$$

where:

P_i = productivity of the i^{th} crop (kg DW/m^2)
Yh_i = harvest yield of i^{th} crop (kg DW)
Ah_i = the area planted to crop i that is harvested (m^2)

Belcher and Travis (1989) have calculated average productivity values for several animal and human foods; these values are presented in Table 5-1. In addition, Shor et al. (1982) present average productivity values of leafy vegetables, exposed produce, protected produce, grain for human consumption, honey, pasture, hay, silage, and animal grain for all U.S. counties. The data in Shor et al. (1982) are presented as fresh weight and can be converted to dry weight using the water content values presented in Baes et al. (1984).

TABLE 5-1

Average Values of Productivity (kg DW/m^2)

Plant Group	Range	Arithmetic Mean
Animal Forage	0.02-0.75	0.31
Animal Grain	0.14-0.45	0.30
Potatoes	0.41-0.56	0.48
Leafy Vegetables	0.09-0.35	0.18
Root Vegetables	0.09-0.44	0.34
Garden Fruits	0.01-0.25	0.09
Legumes	0.08-0.13	0.10

Source: Belcher and Travis, 1989

5.2.3. Plant Pollutant Concentration Due to Air-to-Plant Transfer (Pv_i). In addition to the root uptake and deposition routes, plants can accumulate vapor-phase contaminants by air-to-plant transfer. Air-to-plant transfer is likely to be important for leafy plants and, to a lesser extent, exposed produce. However, this pathway need not be calculated for protected produce.

Two parameters are necessary to estimate air-to-plant transfer: the air-to-plant biotransfer factor and the atmospheric concentration of pollutant at ground level. The latter is the sum of the concentration of pollutant in the vapor phase due to emissions from the combustor and the atmospheric concentration due to volatilization of pollutant deposited on the soil:

$$Pv_i = \frac{[(Fv \cdot Cy) + Cv] \cdot Bv_i}{pa} \qquad \text{(Equation 5-13)}$$

where:

Pv_i = concentration of pollutant due to air-to-plant transfer in the i^{th} plant group (μg pollutant/g plant tissue DW)
Fv = fraction of pollutant air concentration present in the vapor phase (unitless)
Cy = concentration of pollutant in air due to direct emissions (μg pollutant/m^3 air)
Cv = concentration of pollutant in air due to volatilization from soil (μg pollutant/m^3 air)
Bv_i = air-to-plant biotransfer factor for the i^{th} plant group [(μg pollutant/g plant tissue DW)/(μg pollutant/g air)]
pa = density of air (1190 g/m^3 at 25° C)

The example calculations in Section 5.5 demonstrate that Cv can be up to seven or eight orders of magnitude lower than Cy. Thus, Cv will usually make an insignificant contribution to the overall concentration in plants due to air-to-plant transfer.

5.2.3.1 AIR CONCENTRATION DUE TO EMISSIONS (Cy) -- Refer to Chapter 3 to find methods for determining ground level ambient air concentrations averaged over the area surrounding the combustor.

5.2.3.2. FRACTION OF POLLUTANT IN VAPOR PHASE (Fv) -- Using Fv accounts for the fact that organic compounds can exist in the atmosphere in both the vapor and particulate phase, and only the vapor concentration is considered for this pathway. The ratio of vapor to particle depends on the chemical properties of the pollutant and the total suspended particulate concentration (Bidleman, 1988).

Junge (as reported in Bidleman, 1988) developed a model for estimating the adsorbed fraction of contaminants; this model has been modified to determine Fv as shown below:

$$Fv = 1 - [cS_T/(p^\circ_L + cS_T)] \qquad \text{(Equation 5-14)}$$

where:

Fv = fraction of pollutant in vapor phase (unitless)
c = assumed constant of 1.7×10^{-4} (atm-cm)
S_T = Whitby's average total surface area of particles (cm^2/cm^3)
p°_L = liquid-phase vapor pressure (atm)

A detailed discussion of these parameters is presented in Bidleman (1988).

Bidleman (1988) calculated the percentage of particulate for several semivolatile organic compounds as a function of liquid-phase vapor pressure, assuming an S_T of 3.5×10^{-6} cm^2/cm^3 for a background plus local sources particulate situation. Based on Bidleman's calculation, this document assumes that the Fv for benzo(a)pyrene is 0.18. Note that this value is highly uncertain because it is derived from a single calculation, and correlation of the model with field studies is still in progress.

5.2.3.3. AIR-TO-PLANT BIOTRANSFER FACTORS (Bv) -- The air-to-plant biotransfer factor is defined as the ratio of contaminant concentration in aerial plant parts (µg/g DW) to the concentration of pollutant in air (µg/g) (Travis and Hattemer-Frey, 1988). Little is known about the mechanisms of vapor uptake by leaves, and there are few experimentally derived air-to-leaf biotransfer factors available in the literature. Therefore, models to predict Bv based on empirical correlations with chemical properties have been developed. Research on azalea leaf uptake of ten organic vapors has shown that Bv is related to both the Henry's Law constant (H) and the octanol-water partition coefficient (K_{ow}) in the following manner (Bacci et al., 1989):

$$\mathrm{Log}(Bv \cdot H) = -0.93 + 1.14\ \mathrm{Log}K_{ow} \qquad \text{(Equation 5-15)}$$

where:

Bv = air-to-leaf biotransfer ([µg/g DW]/[µg/g])
H = Henry's Law constant (Pa m^3/mol)
K_{ow} = octanol-water partition coefficient (unitless)

Thus, lipophilic compounds with a high Henry's Law constant will tend not to bioaccumulate, but compounds with a high octanol-water coefficient will tend to bioaccumulate (Bacci et al., 1989).

If appropriate measured values for Bv are not available, risk assessors can use Equation 5-15 to predict values for Bv. (Note: Equation 5-15 was derived using H values with the units Pa m^3/mol, and H values with the same units must be used to predict Bv.)

5.2.3.4. AIR CONCENTRATION DUE TO VOLATILIZATION FROM SOIL (Cv) -- Two assumptions were made concerning volatilization of contaminants from soil: the area over which the pollutant volatilizes is essentially circular and the soil emits a uniform amount of pollutant per unit area. Given these assumptions, the air concentration of pollutant is estimated by a gaussian plume formulation for an area source:

$$[Cv] = \sum_{i=1}^{6} F_i \int_0^R \frac{2q}{\sqrt{2\pi}\,\sigma_{z_i} \cdot [\bar{u}]} \cdot \exp\left[-\frac{1}{2}\left(\frac{Ht}{\sigma_{z_i}}\right)^2\right] dx \qquad \text{(Equation 5-16)}$$

where:

Cv = concentration of pollutant in air due to volatilization from soil (µg pollutant/ m^3 air)
F_i = fraction of time of the i^{th} atmospheric stability (unitless)
R = radius of the contaminated area (m)
q = emission rate of pollutant from soil (µg/m^2/s)
Ht = height of plant above soil (m)
[u] = average wind speed above contaminated area (m/s)
σ_{zi} = gaussian plume dispersion parameter for the vertical direction for the i^{th} Pasquill atmospheric stability category (m)

5.2.3.4.1. Fraction of Time of Atmospheric Stability (F_i) -- This parameter represents the percent of time that each of the six Pasquill Stability categories is assumed to occur at a particular site. The Pasquill Stability categories relate the types of turbulence to weather conditions at a site (Slade, 1968). The six categories include A, extremely unstable; B, moderately unstable; C, slightly unstable; D, neutral; E, slightly stable; and F, moderately stable. F_i is determined by collecting meteorologic data at a site for several years. These data are available from the National Weather Service for some locations.

5.2.3.4.2. Radius of Contaminated Area (R) -- This parameter defines the size of the source being considered to emit pollutants. In general, risk assessors may want to choose a value of R that represents the approximate size of the field in which plants being exposed to volatiles from soil are growing. In the example calculations, R is assumed to be 100 m.

5.2.3.4.3. Emission Rate (q) -- The emission rate of pollutant from soil (q) is calculated by:

$$q = Sc \cdot ksv \cdot Z \cdot BD \cdot 3.171\times10^{-4} \quad \text{(Equation 5-17)}$$

where:

q = emission rate of pollutant from soil (μg pollutant/m^2/s)
Sc = soil concentration of pollutant (μg pollutant/g soil)
ksv = soil loss constant due to volatilization (yr^{-1})
Z = soil depth (cm)
BD = soil bulk density (1.5 g/cm^3)
3.171×10^{-4} = conversion factor, 10^4 cm^2/m^2 and 3.171×10^{-8} yr/s

The input values required for calculating emission rate are all discussed in Chapter 4, where default values are presented. A soil concentration based on 20 cm depth is assumed for all plants raised in tilled soil. A soil concentration based on 1 cm is assumed for forage, which is raised in untilled soil.

5.2.3.4.4. Height of Plant (Ht) -- This parameter represents the average height from the ground for the plants under consideration. This methodology assumes a generic plant height of 0.168 meters. Risk assessors may obtain site-specific data from the local extension service.

5.2.3.4.5. Average Wind Speed ([u]) -- Wind speed data are collected by the National Weather Service at several stations around the country. Average U.S. wind speeds range from 2.8 to 6.3 m/s. Wind speed data for selected U.S. stations are presented in Table 5-2. When selecting average wind speed data, assessors should choose data from the closest city that has similar climate and topographical characteristics to the site under consideration.

5.2.3.4.6. Gaussian Plume Dispersion Parameter (σ_{zi}) -- The gaussian plume dispersion parameter (σ_{zi}) is defined as the standard deviation of the distribution of material in a plume in the vertical direction (Slade, 1968) and reflects the ability of

TABLE 5-2
Mean Wind Speed[a] for Selected United States Stations

Station	State	[u] (m/s)	Station	State	[u] (m/s)	Station	State	[u] (m/s)
Birmingham	AL	3.3	Boston	MA	5.6	Columbus	OH	3.9
Montgomery	AL	3.0	Detroit	MI	4.6	Dayton	OH	4.6
Tucson	AZ	3.7	Grand Rapids	MI	4.5	Toledo	OH	4.2
Yuma	AZ	3.5	Lansing	MI	4.6	Oklahoma City	OK	5.7
Fort Smith	AR	3.4	Sault St. Marie	MI	4.3	Tulsa	OK	4.7
Little Rock	AR	3.6	Duluth	MN	5.1	Portland	OR	3.5
Frenso	CA	2.8	Minneapolis	MN	4.7	Harrisburg	PA	3.4
Red Bluff	CA	3.9	Jackson	MS	3.4	Philadelphia	PA	4.3
Sacramento	CA	3.7	Columbia	MO	4.4	Pittsburg	PA	4.2
San Diego	CA	3.0	Kansa City	MO	4.6	Scranton	PA	3.8
Denver	CO	4.1	St. Louis	MO	4.2	Huron	SD	5.3
Grand Junction	CO	3.6	Springfield	MO	5.0	Rapid City	SD	5.0
Pueblo	CO	3.9	Billings	MT	5.1	Chattanooga	TN	3.3
Hartford	CT	4.0	Great Falls	MT	5.9	Knoxville	TN	3.3
Washington	DC	3.4	Havre	MT	4.5	Memphis	TN	4.1
Jackonville	FL	3.8	Helena	MT	3.5	Nashville	TN	3.6
Tampa	FL	3.9	Missoula	MT	2.7	Abilene	TX	5.4
Atlanta	GA	4.1	North Platte	NE	4.6	Amarillo	TX	6.1
Macon	GA	3.5	Omaha	NE	4.8	Austin	TX	4.2
Savannag	GA	3.6	Valentine	NE	4.8	Brownsville	TX	5.3
Boise	ID	4.0	Ely	NV	4.7	Corpus Christi	TX	5.4
Pocatello	ID	4.6	Las Vegas	NV	4.0	Dallas	TX	4.9
Chicago	IL	4.6	Reno	NV	2.9	El Paso	TX	4.2
Moline	IL	4.4	Winnemucca	NV	3.5	Port Authur	TX	4.5
Peoria	IL	4.6	Conconrd	NH	3.0	San Antonio	TX	4.2
Springfield	IL	5.1	Albuquerque	NM	4.0	Salt Lake City	UT	3.9
Evansville	IN	3.7	Roswell	NM	4.1	Burlington	VT	3.9
Fort Wayne	IN	4.6	Albany	NY	4.0	Lynchburg	VA	3.5
Indianapolis	IN	4.3	Binghampton	NY	4.6	Norfolk	VA	4.7
Burlington	IA	4.6	Buffalo	NY	5.5	Richmond	VA	3.4
Des Moines	IA	5.0	New York	NY	5.5	Quillayute	WA	3.0
Sioux City	IA	4.9	Rochester	NY	4.3	Seattle	WA	4.1
Concordia	KS	5.4	Syracuse	NY	4.4	Spokane	WA	3.9
Dodge City	KS	6.3	Cape Hagtteros	NC	5.1	Green Bay	WI	4.6
Topeka	KS	4.6	Charlotte	NC	3.4	Madison	WI	4.4
Wichita	KS	5.6	Greensboro	NC	3.4	Milwaukee	WI	5.3
Louisville	KY	3.8	Wilmington	NC	4.0	Cheyenne	WY	5.9
Shreveport	LA	3.9	Bismarck	ND	4.7	Lander	WY	3.1
Portland	ME	3.9	Fargo	ND	5.7	Sheridan	WY	3.6
Baltimore	MD	4.2	Cleveland	OH	4.8	Elkins	WV	2.8

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[a] Data taken from Local Climatological Data Annual Summaries for 1977. U.S. Department of Commerce, National Oceanic and Atmospheric Administration/Environmental Data Service/National Climatic Data Center.

contaminants to diffuse in the vertical direction. σ_z for each Pasquill stability category can be estimated by the following equation:

$$\sigma_{zi} = a_i(x)^{bi} \qquad \text{(Equation 5-18)}$$

where:

σ_{zi} = gaussian plume dispersion parameter for the i^{th} Pasquill atmospheric stability category (m)
a_i and b_i = empirically determined parameters for each stability category
x = downwind distance from source (m)

The interval $0 < x < R$ is divided into a specified number of smaller segments. This number determines the size of each segment. For example, if 50 segments were specified for a source with an R of 100 meters, the interval would be broken down into 50 segments with a length of 2 meters. The endpoints of each of these segments correspond to an estimate of x. The values of a and b vary according to distance and stability category and are presented in Table 5-3.

5.2.4. Phytotoxicity. Pollutants emitted from combustors may be toxic to plants at certain plant concentrations. If the contaminant is phytotoxic, a maximum tissue concentration for a given crop can be determined based on available phytotoxicity data and assumed as an upper limit to plant uptake. Maximum concentrations are those associated with severe yield reduction (>75%) or plant death, which would preclude pollutant passage up the foodchain. Phytotoxicity may be altered by soil pH (see

TABLE 5-3

Parameters Used to Calculate σ_z

Pasquill Category	x (km)	a	b
A	<0.1	122.8	0.945
	0.10-0.15	158.1	1.05
	0.16-0.20	170.2	1.09
	0.21-0.25	179.5	1.12
	0.26-0.30	217.4	1.26
	0.31-0.40	258.9	1.41
	0.41-0.50	346.8	1.73
	0.51-3.11	453.9	2.12
	>3.11	**	**
B	<0.20	90.7	0.932
	0.21-0.40	98.5	0.983
	>0.40	109.3	1.10
C	All	61.1	0.915
D	<0.30	34.5	0.870
	0.31-1.00	32.1	0.811
	1.01-3.00	32.1	0.644
	3.01-10.0	33.5	0.605
	10.01-30.0	36.7	0.566
	>30.0	44.1	0.512
E	<0.10	24.3	0.837
	0.10-0.30	23.3	0.820
	0.31-1.00	21.6	0.757
	1.01-2.00	21.6	0.631
	2.01-4.00	22.5	0.572
	4.01-10.0	24.7	0.505
	10.01-20.0	27.0	0.467
	20.01-40.0	35.4	0.376
	>40.0	47.6	0.296
F	<0.20	15.2	0.816
	0.21-0.70	14.5	0.784
	0.71-1.00	14.0	0.685
	1.01-2.00	14.0	0.632
	2.01-3.00	14.8	0.545
	3.01-7.00	16.2	0.465
	7.01-15.0	17.8	0.415
	15.01-30.0	22.7	0.327
	30.01-60.0	27.1	0.274
	>60.0	34.2	0.217

Source: U.S. EPA, 1986e

Section 5.2.1.2.1.); the phytotoxicity data chosen should be appropriate for the soil pH of site of the assessment.

5.3. CALCULATING CONCENTRATION OF POLLUTANT IN ANIMAL TISSUES

The animals that humans usually consume as food take up pollutants from the food chain by ingesting plants and ingesting soil while grazing. The food chain model calculates the concentration of pollutant in animal tissues by considering the concentration of pollutant in plants and soil, the quantity of plants and soil that animals consume, and the biotransfer factor of each type of animal tissue:

$$A_j = [\sum_{i=1}^{n}(Qp_{ij} \cdot P_{ij}) + (Qs_j \cdot Sc)] \cdot Ba_j$$

(Equation 5-19)

where:

A_j = concentration of pollutant in the j^{th} animal tissue group (μg pollutant/g animal tissue DW)
Qp_{ij} = quantity of i^{th} plant group eaten by the j^{th} animal each day (kg plant tissue DW/day)
P_{ij} = total concentration of pollutant in the i^{th} plant group eaten by the j^{th} animal (μg pollutant/g plant tissue DW)
Qs_j = quantity of soil eaten by the j^{th} animal each day (kg soil/day)
Sc = soil concentration (μg pollutant/g soil)
Ba_j = biotransfer factor for the j^{th} animal tissue group (d/kg)

Section 5.5. calculates A_j for each of seven animal tissue groups that humans are most likely to consume: beef, beef liver, dairy, pork, poultry, eggs, and lamb. Note that a separate A_j should be calculated for each pollutant to be considered.

5.3.1. Quantity of Plants Consumed by Animals (Qp_{ij}). Animals raised for food are fed a variety of plants including grain (corn, oats, wheat, etc.), forage (pasture grass, hay) and silage. In addition, grazing animals ingest soil that clings to plants. Plant ingestion rates are generally presented as the amount of plant tissue (kg dry weight) consumed in one day.

Most of the available data on plant ingestion rates are limited to cows. Beef cattle have overall ingestion rates ranging from 6-18 kg DW/day (Hoffman and Baes, 1979; Travis and Arms, 1988; McKone and Ryan, 1989) with an average of 12 kg DW/day (McKone and Ryan, 1989). Dairy cattle have overall ingestion rates ranging from 12-25 kg DW/day (Simmonds and Linsley, 1981; Hoffman and Baes, 1979; McKone and Ryan, 1989) with an average of 17 kg DW/day (McKone and Ryan, 1989). Calves consume 8 kg DW/day (NAS, 1987). Boone et al. (1981) have estimated the ingestion rates of seven types of grain, two types of forage, and two types of silage for both beef and dairy cattle. For beef cattle, the average grain, forage and silage consumption is 0.47 kg DW/day, 8.8 kg DW/day and 2.5 kg DW/day, respectively. For dairy cattle, the average grain, forage and silage consumption is 2.6 kg DW/day, 11 kg DW/day and 3.3 kg DW/day, respectively. These average values are used as input variables in the example calculations.

Less information about feed consumption rates is available for other food producing animals. Reported ingestion rates for hogs range from 3.4 (Boone et al., 1981) to 5.2 kg DW/day for lactating sows (NAS, 1987). The average of this range, 4.3 kg DW/day, is used in the example calculations. Based on data in U.S.

EPA (1982b), 30% (1.3 kg DW/day) of the average daily intake for hogs could be silage and the remaining 70% (3.0 kg DW/day) would be grain. Because hogs are not grazing animals, they are assumed not to eat forage.

Reported ingestion rates for chickens range from 0.074 (NAS, 1987) to 0.087 kg DW/day (Boone et al., 1981). The average of this range, 0.08 kg DW/day, is used in the example calculations. This average daily intake is assumed to be all grain because poultry are not grazing animals. Boone et al. (1981) have presented the ingestion rates of seven types of grain for both hogs and chickens, if this information is required for a site-specific assessment.

Ingestion rates for sheep range from 1.2 (NAS, 1987) to 1.6 kg DW/day (Belcher and Travis, 1989). The average of this range, 1.4 kg DW/day, is used in the example calculations. No data concerning the types of feed sheep consume are available. Therefore, sheep are assumed to eat only forage because forage has the highest pollutant concentration of plants eaten by animals.

5.3.2. Quantity of Soil Consumed by Animals (Qs_j). Soil ingestion is of concern for grazing animals, such as cows and sheep, that may ingest plant roots and adhering soil while grazing. Animals could be exposed to pollutants in combustor emissions soon after deposition on soil. In addition, pastures where animals graze are irregularly tilled or untilled, so contaminants are not likely to become incorporated. Therefore, for this pathway, all of the deposited contaminant is assumed to be within the uppermost 1-cm soil layer, and ingested soil is assumed to originate from the same 1-cm layer.

Soil ingestion rates for animals are usually presented as a percent of the daily food intake. Reported soil ingestion rates for cattle are 3-6% (Simmonds and Linsley, 1981) and 1-18% (Thornton and Abrams, 1983) of dry matter intake. Resulting daily soil intake for cattle could range from 0.17 to 3.1 kg/day based on an average dry matter intake of 17 kg/day in dairy cattle. McKone and Ryan (1989) report daily soil intake for cattle ranging from 0.1 to 0.7 kg/day with an average of 0.4 kg/day. Thornton and Abrams (1983) note seasonal variation in soil ingestion rates; soil ingestion is highest in early spring when there is limited forage. In this document, approximately 3% of the forage intake for beef and dairy cattle will be used in the example calculations for the quantity of soil consumed.

Reported soil ingestion rates for sheep are 20% (Simmonds and Linsley, 1981) and up to 30% for sheep during winter months when forage is reduced (Thornton and Abrams, 1983). Resulting daily soil intake for sheep is 0.28-0.42 kg/day based on an average dry matter intake of 1.4 kg/day. In this document, approximately 20% of the forage intake for sheep will be used in the example calculations for the quantity of soil consumed.

Although hogs and poultry are not grazing animals, these animals are usually raised outdoors and will undoubtedly ingest some soil when they are raised for home consumption. However, this document will assume no soil ingestion for hogs or poultry because appropriate data are not available.

5.3.3. Biotransfer Factors (Ba_j). Biotransfer factors are defined as the ratio of pollutant concentration in animal tissue to the daily intake of pollutant by the animal. These factors are a measure of the degree to which contaminants are transferred from the environment (food and water) to specific animal tissues; as such, they are chemical and tissue specific. Transfer factors for several inorganic compounds in beef, pork, lamb, poultry, and eggs are presented in Ng et al. (1982) and summarized in Table 5-4.

Often uptake data in animal tissue are presented in the literature as the ratio of contaminant concentration in animal tissue to contaminant concentration in feed ([μg contaminant/kg tissue DW]/[μg contaminant/kg food DW]). The biotransfer factor can be determined by dividing the concentration ratio by the animal feed ingestion rate (kg food DW/day) (Ng et al., 1982; Belcher and Travis, 1989). If literature data are presented as wet weight,they can be adjusted using water content value of 0.6 for beef muscle and 0.9 for cows milk (Baes et al., 1984). Water content values are also presented in U.S. Department of Agriculture (1975).

The biotransfer factor of an organic compound is directly proportional to the compound's octanol-water partition coefficient, K_{ow} (Travis and Arms, 1988). Therefore, the K_{ow} can be used to estimate biotransfer factors for organic compounds if experimental

TABLE 5-4

Biotransfer Factors for Selected Metals (Ba_j)

Metal	Animal Tissue	Biotransfer Factor (d/kg)
Cadmium[a]	Beef	0 to 0.12
	Beef liver	0.045 to 0.379
	Lamb	0.003 to 0.06
	Pork	0 to 0.12
	Poultry	1.36 to 36.57
	Dairy	0.0036 to 0.0076
	Eggs	0.489 to 0.85
Copper[b]	Beef	0.005 to 0.012
	Pork	0.014 to 0.029
	Lamb	0 .039
	Poultry	0.51
	Eggs[c]	0.72
Iron[b]	Beef	0.012 to 0.024
	Pork	0.026
	Lamb	0.073
	Poultry	1.5
	Eggs[c]	2.0
Lead[b]	Beef	0.0007
Magnesium[b]	Beef	0.023
Manganese[b]	Beef	0.0006
	Pork	0.002 to 0.005
	Lamb	0.002 to 0.012
	Poultry	0.01 to 0.068
	Eggs[c]	0.031 to 0.13
Selenium[b]	Pork	0.32
	Poultry	1.5 to 6.8
	Eggs[c]	2.0 to 7.5
Zinc[b]	Beef	0.035 to 0.2
	Pork	0.14 to 0.24
	Lamb	4.1
	Poultry	6.5
	Eggs[c]	4.6

---

[a] Source: Belcher and Travis, 1989
[b] Source: Ng et al., 1982
[c] Units for eggs are d/kg egg contents

data are not available. The relationship between K_{ow} and biotransfer factor for beef and milk can be expressed as:

$$\log Ba(\text{beef}) = -7.6 + \log K_{ow} \qquad \text{(Equation 5-20)}$$

and

$$\log Ba(\text{milk}) = -8.1 + \log K_{ow} \qquad \text{(Equation 5-21)}$$

Similar equations for other animal tissues are not available and the applicability of these equations to other types of animal tissues is not known.

5.4. CALCULATING HUMAN DAILY INTAKE

Human daily pollutant intake from consumption of contaminated plants is calculated by multiplying the concentration of pollutant in each plant group by the amount of contaminated plant group consumed daily. Similarly, daily pollutant intake from the consumption of meat, dairy products, or eggs from animals that have fed upon contaminated forage, grain or soil is determined from the concentration of pollutant in the animal tissue and the amount of each contaminated animal tissue that is consumed daily.

5.4.1. Calculating Daily Intake from Contaminated Plants. The human daily intake of pollutants due to consumption of a specific plant group is calculated by:

$$Ip_i = P_i \cdot Fp_i \cdot Cp_i \qquad \text{(Equation 5-22)}$$

where:

Ip_i = human daily intake of pollutant due to consumption of the i^{th} plant group (μg pollutant/kg BW/day)
P_i = total concentration of pollutant in the i^{th} plant group (μg pollutant/g plant tissue DW)
Fp_i = fraction of i^{th} plant group assumed to originate from contaminated soil (unitless)
Cp_i = human daily dietary consumption of the i^{th} plant group for the appropriate age group (g plant tissue DW/kg BW/day)

Total daily intake from consumption of all plant groups is calculated by summing the contributions of each individual plant group:

$$Ip = \sum_{i=1}^{n} Ip_i \qquad \text{(Equation 5-23)}$$

where:

Ip = total daily intake from consumption of plant tissue (μg pollutant/kg BW/day)
Ip_i = daily intake from consumption of i^{th} plant group (μg pollutant/kg BW/day)

5.4.1.1. PLANT GROUP CONSUMPTION RATES (Cp_i) -- An up-to-date and detailed source of information regarding food consumption habits of the U.S. population may be found in the U.S. EPA Office of Pesticide Programs' "Tolerance Assessment System (TAS)". A specific analysis conducted by Technical Assessment Systems, Inc. (U.S. EPA, 1988d) for the U.S. EPA using the TAS is of particular relevance to this methodology because the results are converted to a dry rather than wet weight basis. Also, the data are presented as frequency distributions of consumption instead of as population averages. This analysis determined the distribution of mean, individual daily consumption, in g DW food/kg BW, of seven plant food groups for the total U.S. population and five subgroups: males

> 13 years, females > 13 years, infants < 1 year, children 1-6 years, and children 7-12. The results are presented as a frequency distribution in equal intervals based on population percentiles.

The seven food groups include grains and cereals; potatoes, including sweet potatoes; leafy vegetables, excluding brassica; brassica vegetables; legumes; fruiting vegetables (e.g., tomatoes, cucumbers, squash, melons, etc.) and root vegetables. A complete description of each food group is presented in Table 5-5. These seven groups can be placed into the three broad plant categories (leafy, exposed, protected) when calculating plant pollutant concentration in the following manner. Foods in the leafy vegetables and brassica vegetables food groups are considered to be leafy produce. Foods in the grains, potatoes, legumes and root vegetables food groups are considered to be protected produce. In addition, melons, which are in the fruiting vegetables food group, are also considered to be protected produce. The remaining foods in the fruiting vegetables group as well as fruits are considered to be exposed produce.

The consumption data used in the U.S. EPA (1988d) analysis were collected for the U.S. Department of Agriculture Nationwide Food Consumption Survey (NFCS) of 1977-78 (U.S. Department of Agriculture, 1983). Three-day food consumption records were collected during the period of April 1977 through March 1978. About 15,000 households representing 30,770 individuals were surveyed. The NFCS has the advantage of a large

TABLE 5-5

Description of Seven Plant Food Groups

Food Group	Food Name	Food Group	Food Name
Grains	barley		
	corn -	Potatoes	white -
	bran		NS*
	sugar		peeled
	endosperm		dry
	popcorn		peel only
	sweet corn		whole
	oil		sweet potatoes
	oats		
	rice -	Brassica	broccoli
	milled		brussel sprouts
	rough		cabbage
	rye -		cauliflower
	germ		collards
	flour		kale
	rough		kohlrabi
	sorghum		mustard greens
	wheat -		
	germ		
	bran		
	flour		
	rough		
	millet		
	buckwheat		
Leafy Vegetables	beet tops	Root Vegetables	carrots
	celery		taro root
	chicory		garlic
	lettuce -		artichokes
	leafy		leeks
	NS*		onions
	head		radishes
	cress -		rutabagas
	garden		salsify
	upland		shallots
	field		turnips
	dandelion		parsnips
	parsley		cassava
	rhubarb		beets
	spinach		
	swiss chard		
	turnip tops		

TABLE 5-5 (cont.)

Food Group	Food Name	Food Group	Food Name
Leafy, cont.	taro greens		
	endive		
	radish tops		
	rutabaga tops		
Legumes	dry beans -	Fruiting Vegetables	
	great northern		
	kidney		paprika
	lima		cantaloupe
	navy		casaba
	pinto		crenshaw
	broadbeans		honey dew
	pigeon		persian melon
	hyacinth		watermelon
	blackeye peas		cucumber
	garbanzo		pumpkin
	succulent beans -		summer squash
	lima		winter squash
	green		eggplant
	yellow/wax		peppers -
	broadbeans		sweet
	peas, dry and green		chili
	lentils, whole and split		other
	sprouts, mung and soy bean		tomatoes -
	carob		whole
	soybeans -		juice
	oil		puree
	mature		paste
	flour		catsup
	NS*		

* NS = not specified
Source: U.S. EPA (1988d)

sample size, and it has been weighted to be statistically representative of the U.S. population including race, age, sex, regions and seasons (U.S. EPA, 1988d).

The major limitation of the NFCS for exposure assessment is that consumption is expressed in terms of foods "as eaten" (pizza, apple pie, french toast) rather than in terms of the commodities for which residue data are most often collected (wheat, beef, apples) (U.S. EPA, 1988d). The U.S. EPA (1989b) has adapted the NFCS so that it can be used to estimate the consumption of raw agricultural commodities. A total of 3734 foods were examined and the following information was entered into a master file: components that made up the food; the percentage, by weight, of each component; the reference source for the formula used to estimate food composition in terms of components; and the form of each component as it was eaten (raw, cooked, fried, etc.). The TAS analysis is based on the data entered into this master file, and the mean daily consumption rates of the seven food groups predicted by TAS are presented in Table 5-6.

The U.S. EPA (1988d) analysis included fruiting vegetables, but not fruits per se (e.g., apples, strawberries, cherries, etc.). The consumption rates for fruits were developed instead from information presented in Appendix A of U.S. EPA (1989b). The fruit data were grouped, and the total average dry-weight consumption for children (2 years, 13.5 kg BW) and adults (males and females ≥14, 70 kg BW) was calculated. Coefficients of variation for consumption of fruits and vegetables (child=0.0955, adult=0.04) were calculated from data in McKone and Ryan (1989). Consumption

TABLE 5-6
Consumption Rate of Seven Plant Groups (Cp_i)
(g DW/kg BW/day)

Plant Group	U.S. Population	Males >13 yrs	Females >13 yrs	Children 7-12 yrs	Children 1-6 yrs	Infants <1 yr
90-95th Percentile						
Grains	4.4	3.0	2.3	4.9	6.9	7.7
Potatoes	0.72	0.58	0.50	0.91	1.3	1.2
Leafy	0.06	0.05	0.06	0.06	0.06	0.008
Brassica	0.07	0.06	0.07	0.08	0.09	0
Legumes	1.4	1.0	0.89	1.7	2.3	4.6
Fruiting	0.30	0.24	0.24	0.38	0.20	0.32
Root	0.11	0.08	0.09	0.13	0.20	0.63
Fruits[a]		0.34[b]			2.19[c]	
70-75th Percentile						
Grains	2.4	2.0	1.5	3.6	5.0	4.6
Potatoes	0.34	0.31	0.26	0.49	0.65	0.25
Leafy	0.02	0.02	0.02	0.02	0.02	0
Brassica	0	0	0.001	0	0	0
Legumes	0.67	0.57	0.49	0.98	1.3	2.4
Fruiting	0.13	0.11	0.10	0.19	0.24	0.07
Root	0.04	0.03	0.03	0.05	0.07	0.26
Fruits[a]		0.33[b]			2.08[c]	
50-55th Percentile						
Grains	1.6	1.5	1.2	2.9	4.1	3.0
Potatoes	0.19	0.19	0.15	0.30	0.39	0.07
Leafy	0.008	0.008	0.009	0.009	0.004	0
Brassica	0	0	0	0	0	0
Legumes	0.44	0.40	0.34	0.69	0.87	1.1
Fruiting	0.07	0.06	0.05	0.11	0.13	0.01
Root	0.02	0.02	0.02	0.03	0.03	0.08
Fruits[a]		0.32[b]			1.95[c]	

[a] Calculated from data presented in U.S. EPA (1989b) and McKone and Ryan (1989), see text.

[b] Average of males and females ≥14 years

[c] Data for 2-year-old children only

Source (except fruits): U.S. EPA, 1988d

was assumed to be normally distributed over the U.S. population, and the above coefficients of variation were used with "Z-scores" of 0.68 and 1.28 (corresponding to areas under the unit normal distribution of 75% and 90% of the total area, respectively) to estimate 70-75th and 90-95th percentiles of fruit consumption. These values are presented in Table 5-6.

5.4.1.2. FRACTION ASSUMED TO ORIGINATE FROM CONTAMINATED SOIL (Fp_i) -- The fraction of plant foods produced on contaminated soil (at home) is primarily influenced by the number of households gardening and the size of home garden. A total of 38% of U.S. households raised a home garden in 1986, and the average garden was 325 square feet in 1986 (U.S. EPA, 1989a). It will be assumed that home gardeners may produce and consume leafy, legume and root vegetables; potatoes; fruiting vegetables; fruits and some grains. The most commonly raised garden vegetables are tomatoes, peppers, onions, cucumbers, beans and lettuce (U.S. EPA, 1989a). Fp_i was calculated from the 1977-78 NFCS and values are presented in Table 5-7. For foods in the "General" category on Table 5-7, Fp_i was calculated by dividing the reported weekly quantities of foods produced at home by the quantities of foods from all sources. For the specific food categories on Table 5-7, the quantities produced at home were not reported. Therefore, the reported quantities of bought foods were subtracted from the reported quantities of food from all sources, and this difference was divided by the quantities of food from all sources to determine the percentage "produced at

TABLE 5-7
Percentage of Some Plant Foods Produced At Home (Fp_i)*

Food Groups	All Urbanizations	Central City	Suburban	Non-Metropolitan
General				
Flour, cereal	0.1	0.1	0.2	0.2
Sugar, sweets	1.5	0.5	1.6	2.6
Potatoes	8.6	1.5	4.6	17.2
Vegetables (fresh, frozen, canned)	15.9	5.8	14.0	27.0
Fruit (fresh, frozen, canned)	5.7	2.0	5.5	9.8
Juice (vegetable, fruit)	1.8	0.3	1.3	3.8
Dried fruit, vegetables	3.0	1.3	4.0	7.7
Potatoes				
potatoes, white	11.8	3.01	6.54	22.0
sweet potatoes	17.6	9.52	15.4	33.3
Leafy Vegetables				
celery	4.3	4.3	0.0	5.3
lettuce, unspec.	4.2	2.59	4.44	5.83
spinach	20.0	0.0	16.7	33.3
rhubarb	75.0	50.0	50.0	83.3
Brassica Vegetables				
broccoli	6.7	5.88	5.26	20.0
cabbage	16.2	6.49	14.3	27.1
cauliflower	11.1	0.0	9.09	16.7
collards	28.6	15.4	33.3	66.7
kale	0.0	0.0	0.0	50.0
mustard greens	28.6	15.4	33.3	60.0
Legumes				
beans, lima	75.0	50.0	66.7	85.7
beans, succulent	65.9	34.6	57.6	85.7
peas, garden green	72.7	50.0	62.5	80.0
Fruiting Vegetables				
cantaloupe	14.1	5.08	10.8	25.0
other melons	18.1	7.89	17.0	30.5
cucumbers	39.5	21.2	39.0	59.0
pumpkin, winter squash	52.9	38.5	45.0	70.6
peppers	27.8	15.0	26.3	42.9
tomatoes	48.7	31.7	46.6	62.3

TABLE 5-7 (cont.)

Food Groups	All Urbanizations	Central City	Suburban	Non-Metropolitan
Root Vegetables				
beets	85.7	50.0	83.3	90.0
carrots	15.9	7.32	14.3	26.8
onions, dry bulb	10.0	1.75	8.00	19.5
turnips	50.0	0.0	33.3	60.0
Grains				
corn, sweet	44.8	19.5	34.4	66.7
Other Fruit				
oranges	9.3	4.9	23.6	12.0
strawberries	33.3	12.5	26.7	41.7
apples	16.5	8.0	15.8	23.9
peaches	29.7	16.1	30.0	40.0
pears	28.6	14.3	21.7	42.1
grapes	11.8	5.3	5.3	15.4
plums	20.0	18.2	25.0	37.5

*Source: Calculated from U.S. Department of Agriculture, 1983 home". However, Fp_i for the specific food categories may also include foods received as gifts or eaten while the responder was a guest. Thus, the Fp_i values in these categories may be higher than those for foods in the "General" category. For the example calculations, Fp_i for each food category was determined by averaging the values of each individual food within the category.

On Table 5-7 the classes of urbanization are defined as follows. Central city refers to those people who live within the city limits of a Standard Metropolitan Statistical Area (SMSA). Suburban refers to those people who live in a SMSA but do not live within the city limits. Non-metropolitan refers to those people who live outside a SMSA (U.S. Department of Agriculture, 1983).

5.4.2. Calculating Daily Intake from Animal Tissue. Human daily intake due to ingestion of a single contaminated animal tissue is calculated by:

$$Ia_j = A_j \cdot Fa_j \cdot Ca_j \qquad \text{(Equation 5-24)}$$

where:

Ia_j = total daily intake from ingestion of j^{th} animal tissue group (µg pollutant/kg BW/day)
A_j = concentration of pollutant in the j^{th} animal tissue group (µg pollutant/g animal tissue DW)
Fa_j = fraction of the j^{th} animal tissue group assumed to originate from contaminated soil (unitless)
Ca_j = daily human consumption of the j^{th} animal tissue group (g animal tissue DW/kg BW/day)

Total human daily intake from consumption of all animal groups is calculated by summing the contributions of each specific animal group:

$$Ia = \sum_{j=1}^{n} Ia_i \qquad \text{(Equation 5-25)}$$

where:

Ia = total daily intake from consumption of animal tissue (µg pollutant/kg BW/day)
Ia_j = daily intake from consumption of j^{th} animal group (µg pollutant/kg BW/day)

5.4.2.1. ANIMAL TISSUE CONSUMPTION RATES (Ca_j) -- The consumption rates for meat and dairy products were developed from data presented in U.S. EPA (1989b) in the same manner as described for fruits in Section 5.4.1.1. For each meat group and for dairy products, the average dry-weight consumption of muscle tissue (or milk solids) and fat were added. The data for children 2 years old were divided by a body weight of 13.5 kg (Nelson et al., 1969). The data for males and females ≥14 years old were averaged and divided by a body weight of 70 kg.

A coefficient of variation for meat (child=0.5583, adult=0.0326) and a weighted-average coefficient of variation for dairy (child=0.0669, adult=0.0244) were calculated from data in McKone and Ryan (1989). These values were used with z-scores of 0.68 and 1.28 to approximate the 70-75th and 90-95th percentile meat and dairy consumption. These values are presented in Table 5-8.

5.4.2.2. FRACTION OF ANIMAL TISSUE ORIGINATING FROM CONTAMINATED SOIL (Fa_j) -- The model presented in this methodology allows assessors to consider both animals raised at home for home consumption and animals raised commercially in the vicinity of a

TABLE 5-8

Consumption Rates of Meat and Dairy Products (Ca_j)
(g DW/kg BW/day)[a]

Food	Adult[b]	Child[c]
	90-95th Percentile	
Beef	0.5652	2.1290
Beef liver	0.0166	0.0403
Lamb	0.00566	0.0206
Pork	0.3439	1.646
Poultry	0.1176	0.6053
Dairy	0.7432	4.1270
Eggs	0.1250	0.9119
	70-75th Percentile	
Beef	0.5546	1.7130
Beef liver	0.0163	0.0324
Lamb	0.00556	0.0166
Pork	0.3375	1.3245
Poultry	0.1154	0.4870
Dairy	0.7327	3.9745
Eggs	0.1227	0.7337
	50th Percentile	
Beef	0.5425	1.2417
Beef liver	0.0160	0.0235
Lamb	0.00544	0.0120
Pork	0.3301	0.9600
Poultry	0.1129	0.3530
Dairy	0.7207	3.8015
Eggs	0.1200	0.5318

[a] Calculated from data presented in U.S. EPA (1989b) and McKone and Ryan (1989), see text.

[b] Average of males and females ≥14 years

[c] Data for 2-year-old children only

combustor that are distributed locally. Information on the fraction of animal consumption that is home produced is presented in the 1977-78 NFCS (U.S. Department of Agriculture, 1983). Values of Fa_j for home produced animal foods are presented in Table 5-9 and are calculated as discussed in Section 5.4.1.2. The specific values for beef, pork, and poultry were used to calculate human daily intake of benzo(a)pyrene and cadmium. For the remaining meats (beef liver and lamb), the "general meat" value was used because specific meat values were not available. The "general" values for dairy and eggs were also used in the example calculations. However, information on the fraction of commercial animal products that are distributed locally is not available nationally. If this information is important for a site-specific scenario, assessors should use local data.

5.5. EXAMPLE CALCULATIONS

This section illustrates a food-chain exposure to benzo(a)pyrene and cadmium emitted from a combustor for Scenario B. As discussed in Chapter 2, Scenario B is represented by a child who grows up within 5 km of the combustor and remains in the area for 30 years. A portion of all foods are assumed to be produced at home; the rates for suburban households (Tables 5-7 and 5-9) are used to estimate the amount of home-grown food. The amounts of vegetables and meats not produced at home are assumed to be imported into the area and do not contribute to exposure because they are not contaminated. The amount of milk not produced at home

TABLE 5-9

Percentage of Animal Foods Produced At Home (Fa_j)*

Food Groups	All Urbanizations	Central City	Suburban	Non-Metropolitan
General				
Dairy (milk,cream, cheese, etc.)	1.2	0.0	0.3	3.1
meat	4.0	0.2	2.0	9.7
poultry, fish	6.8	1.9	5.9	10.8
eggs	3.3	0.3	2.2	7.9
Beef	7.0	1.4	3.9	15.3
Pork	4.8	1.4	3.4	9.0
fresh	4.7	0.6	2.8	10.7
cured	5.1	1.9	4.2	6.9
Poultry	4.3	1.6	3.6	8.0
chicken	2.6	0.7	1.4	6.0
turkey, other	14.0	8.3	13.0	20.0

*Source: Calculated from U.S. Department of Agriculture, 1983 is assumed to be commercially produced from an area within 50 km of the combustor and, therefore, does contribute to exposure.

The estimataed 70-75th percentile of plant consumption rates (Table 5-6) is used to estimate exposure through plant foods. The rates for children age 1-6 were used to estimate the individual's exposure as a child, and the average rates of males and females ≥14 were used to estimate the individual's exposure as an adult. The 70-75th percentile of meat consumption rates is used to estimate exposure through meats and animal products. These values were calculated as described in Section 5.4.2.1.

5.5.1. Benzo(a)pyrene. This section presents examples of calculating benzo(a)pyrene concentration in plants and animals and calculating total daily intake of benzo(a)pyrene by humans. The input variables used for calculating exposure to benzo(a)pyrene are presented in Table 5-10.

5.5.1.1. CALCULATING BENZO(a)PYRENE CONCENTRATION IN PLANTS -- Exposure through the following eight plant groups is illustrated: forage, grain, potatoes, leafy vegetables (includes brassica), root vegetables, fruit, fruiting vegetables and legumes. Pollutant concentration in these plants is the sum of concentration due to root uptake, direct deposition of particles onto exposed surfaces, and air-to-plant transfer of volatile compounds.

TABLE 5-10

Input Variables For Scenario B Exposure Calculations: B(a)P

Input Variable	Value	Document Section
Sc (μg/g)		4.4.2.
5 km: 20 cm	3.81×10^{-6}	
1 cm	7.66×10^{-5}	
50 km: 20 cm	2.70×10^{-7}	
1 cm	5.43×10^{-6}	
Br_i [(μg/g plant DW)/(μg/g soil)]		5.2.1.2.2.[a]
grains	0.003	
legumes	0.51	
potatoes	0.51	
root vegetables	0.51	
fruits	0.067	
fruiting vegetables	0.067	
leafy vegetables	0.38	
forage	0.38	
Dyd ($g/m^2/yr$)		3.6.
5 km	2.45×10^{-7}	
50 km	3.77×10^{-8}	
Dyw ($g/m^2/yr$)		3.6.
5 km	6.74×10^{-7}	
50 km	2.74×10^{-8}	
Fw (unitless)	0.02	5.2.2.2.[b]
Rp_i (unitless)		5.2.2.3.[c]
grains	0.0	
legumes	0.008	
potatoes	0.0	
root vegetables	0.0	
fruits	0.05	
fruiting vegetables	0.05	
leafy vegetables	0.16	
forage	0.47	
kp (yr^{-1})	126.5	5.2.2.4.
Tp_i (yr)		5.2.2.5.
forage	0.123	
other plants	0.164	

TABLE 5-10 (cont.)

Input Variable	Value	Document Section
Yp_i (kg DW/m^2)		5.2.2.6.[d]
grains	0.22	
legumes	0.43	
potatoes	0.43	
root vegetables	0.43	
fruits	0.26	
fruiting vegetables	0.26	
leafy vegetables	0.14	
forage	0.31	
Bv_i [(μg/g plant DW)/(μg/g air)]		5.2.3.3.[e]
grains	0.0	
legumes	6.1×10^{6}	
potatoes	0.0	
root vegetables	0.0	
fruits	6.1×10^{6}	
fruiting vegetables	6.1×10^{6}	
leafy vegetables	6.1×10^{6}	
forage	6.1×10^{6}	
Cy (μg/m^3)		3.6.
5 km	3.32×10^{-6}	
50 km	6.10×10^{-7}	
Cv (μg/m^3)		5.2.3.4.
5 km: 20 cm soil depth	1.45×10^{-14}	
1 cm soil depth	2.92×10^{-13}	
50 km: 20 cm soil depth	1.03×10^{-15}	
1 cm soil depth	2.07×10^{-14}	
Fv (unitless)	0.18	5.2.3.2.
pa (g/m^3)	1190	
Qs_j (kg DW/day)		5.3.2.
beef	0.3	
beef liver	0.3	
dairy	0.4	
pork	0	
poultry	0	
eggs	0	
lamb	0.4	

TABLE 5-10 (cont.)

Input Variable	Value	Document Section
Qp_{ij} (kg DW/day)		5.3.1.
beef-grain	0.47	
beef-forage	8.8	
beef-silage	2.5	
beef liver-grain	0.47	
beef liver-forage	8.8	
beef liver-silage	2.5	
dairy-grain	2.6	
dairy-forage	11.0	
dairy-silage	3.3	
pork-grain	3.0	
pork-silage	1.3	
poultry	0.08	
eggs	0.08	
lamb	1.4	
Ba_j (d/kg)		5.3.3.[f]
beef	0.027	
beef liver	0.036	
dairy	0.004	
pork	0.019	
poultry	0.031	
eggs	0.049	
lamb	0.020	
Fp_i (unitless)		5.4.1.2.[g]
grains	0.344	
legumes	0.623	
potatoes	0.110	
root vegetables	0.347	
fruits	0.212	
fruiting vegetables	0.308	
leafy vegetables	0.178	
Fa_j (unitless)		5.4.2.2.[h]
beef	0.039	
beef liver	0.020	
dairy	0.003	
pork	0.034	
poultry	0.036	
eggs	0.022	
lamb	0.020	

TABLE 5-10 (cont.)

Input Variable	Value		Document Section
Cp_i (g DW/kg BW/day)			5.4.1.1.
	Children	Adults[i]	
grains	5.0	1.75	
legumes	1.3	0.530	
potatoes	0.65	0.285	
root vegetables	0.07	0.03	
fruits	2.08	0.33	
fruiting vegetables	0.24	0.105	
leafy vegetables	0.02	0.02	
Ca_j (g DW/kg BW/day)			5.4.2.1.
	Children	Adults	
beef	1.713	0.555	
beef liver	0.032	0.016	
dairy	3.975	0.733	
pork	1.325	0.338	
poultry	0.487	0.115	
eggs	0.734	0.123	
lamb	0.017	0.006	

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[a] Source: Risk Assessment of the Dickerson Site (1988) as cited in Belcher and Travis (1989)

[b] Source: Risk Assessment of the Dickerson Site (1988) as cited in Belcher (1989)

[c] Sources: Baes et al. (1984), Hoffman and Baes (1979) and Shor et al. (1982) as cited in Belcher and Travis (1989)

[d] Source: For forage only, Hoffman and Baes (1979) as cited in Belcher and Travis (1989). All other plant groups, Shor et al. (1982) data for Hillsboro County, Florida. Adjusted to DW using water content data in Baes et al. (1984)

[e] Calculated using Equation 5-15; $H = 0.157$ Pa-m^3/mol, $\log K_{ow} = 6.06$ (U.S. EPA, 1986b)

[f] Source: Belcher and Travis (1989)

[g] Table 5-7, average of individual foods within each plant group for suburban households

[h] Table 5-9

[i] Table 5-6, average of 70-75th percentile of males and females >13 years old

Benzo(a)pyrene concentration in plants due to root uptake (Pr) is calculated according to Equation 5-2, assuming a soil depth of 20 cm.

For grains:

$$\begin{aligned} Pr &= 3.81\text{x}10^{-6}\ \mu g/g \cdot 0.003\ (\mu g/g \text{ plant DW})/(\mu g/g \text{ soil}) \\ &= 1.14\text{x}10^{-8}\ \mu g \text{ B(a)P/g plant tissue} \end{aligned}$$

For legumes, potatoes, and root vegetables:

$$\begin{aligned} Pr &= 3.81\text{x}10^{-6}\ \mu g/g \cdot 0.51\ (\mu g/g \text{ plant DW})/(\mu g/g \text{ soil}) \\ &= 1.94\text{x}10^{-6}\ \mu g \text{ B(a)P/g plant tissue} \end{aligned}$$

For fruits and fruiting vegetables:

$$\begin{aligned} Pr &= 3.81\text{x}10^{-6}\ \mu g/g \cdot 0.067\ (\mu g/g \text{ plant DW})/(\mu g/g \text{ soil}) \\ &= 2.55\text{x}10^{-7}\ \mu g \text{ B(a)P/g plant tissue} \end{aligned}$$

For leafy vegetables and forage:

$$\begin{aligned} Pr &= 3.81\text{x}10^{-6}\ \mu g/g \cdot 0.38\ (\mu g/g \text{ plant DW})/(\mu g/g \text{ soil}) \\ &= 1.45\text{x}10^{-6}\ \mu g \text{ B(a)P/g plant tissue} \end{aligned}$$

Benzo(a)pyrene concentration in plants due to direct deposition (Pd) is calculated according to Equation 5-4. Pd is not calculated for grains, potatoes, and root vegetables because they are "protected produce" and are not exposed to direct deposition.

For legumes:

$$Pd = \frac{1000 \cdot [2.45\text{x}10^{-7} + (0.02 \cdot 6.74\text{x}10^{-7})] \cdot 0.008 \cdot [1-\exp(-126.5 \cdot 0.164)]}{(0.43)\ (126.5)}$$

$$= 3.80\text{x}10^{-8}\ \mu g \text{ B(a)P/g plant tissue}$$

For fruits and fruiting vegetables:

$$Pd = \frac{1000 \cdot [2.45\text{x}10^{-7} + (0.02 \cdot 6.74\text{x}10^{-7})] \cdot 0.05 \cdot [1-\exp(-126.5 \cdot 0.164)]}{(0.26)\ (126.5)}$$

$= 3.93\text{x}10^{-7}$ µg B(a)P/g plant tissue

For leafy vegetables:

$$Pd = \frac{1000 \cdot [2.45\text{x}10^{-7} + (0.02 \cdot 6.74\text{x}10^{-7})] \cdot 0.16 \cdot [1-\exp(-126.5 \cdot 0.164)]}{(0.14)\ (126.5)}$$

$= 2.34\text{x}10^{-6}$ µg B(a)P/g plant tissue

For forage:

$$Pd = \frac{1000 \cdot [2.45\text{x}10^{-7} + (0.02 \cdot 6.74\text{x}10^{-7})] \cdot 0.47 \cdot [1-\exp(-126.5 \cdot 0.123)]}{(0.31)\ (126.5)}$$

$= 3.10\text{x}10^{-6}$ µg B(a)P/g plant tissue

Benzo(a)pyrene concentration due to air-to-plant transfer (Pv) is calculated according to Equation 5-13. Pv is not calculated for grains, potatoes, and root vegetables because they are "protected produce" and are not directly exposed to airborne volatile compounds. Theoretically, one would use Cv calculated from a soil concentration at 1 cm depth ($2.92\text{x}10^{-13}$) to calculate Pv for forage. However, since Cv is so small compared to Cy, using a Cv for 1 cm will not cause the Pv for forage to be any different than Pv for other plant groups. Because the inputs for this pathway are chemical specific and not species specific, Pv is the same for each of the remaining plant groups.

$$Pv = \frac{[(0.18 \cdot 3.32\text{x}10^{-6}\ \mu g/m^3) + 1.45\text{x}10^{-14}\ \mu g/m^3] \cdot 6.1\text{x}10^{6}}{1190\ g/m^3}$$

$$= 3.06\text{x}10^{-3}\ \mu g\ B(a)P/g \text{ plant tissue}$$

The total benzo(a)pyrene concentration in plants (P_i) is calculated by adding the pollutant concentrations due to root uptake, direct deposition, and air-to-plant transfer (Equation 5-1).

For grains:

$$P_i = 1.14\text{x}10^{-8}\ \mu g\ B(a)P/g \text{ tissue}$$

For legumes:

$$P_i = 1.94\text{x}10^{-6} + 3.80\text{x}10^{-8} + 3.06\text{x}10^{-3}$$
$$= 3.06\text{x}10^{-3}\ \mu g\ B(a)P/g \text{ tissue}$$

For potatoes:

$$P_i = 1.94\text{x}10^{-6}\ \mu g\ B(a)P/g \text{ tissue}$$

For root vegetables:

$$P_i = 1.94\text{x}10^{-6}\ \mu g\ B(a)P/g \text{ tissue}$$

For fruits and fruiting vegetables:

$$P_i = 2.55\text{x}10^{-7} + 3.93\text{x}10^{-7} + 3.06\text{x}10^{-3}$$
$$= 3.06\text{x}10^{-3}\ \mu g\ B(a)P/g \text{ tissue}$$

For leafy vegetables:

$$P_i = 1.45\text{x}10^{-6} + 2.34\text{x}10^{-6} + 3.06\text{x}10^{-3}$$
$$= 3.06\text{x}10^{-3}\ \mu g\ B(a)P/g \text{ tissue}$$

For forage:

$$P_i = 1.45\text{x}10^{-6} + 3.10\text{x}10^{-6} + 3.06\text{x}10^{-3}$$
$$= 3.06\text{x}10^{-3}\ \mu g\ B(a)P/g \text{ tissue}$$

5.5.1.2. CALCULATING POLLUTANT CONCENTRATION IN ANIMAL TISSUES -- Exposure through the following seven animal food groups is illustrated: beef, beef liver, dairy, pork, lamb, poultry, and eggs. Animals ingest pollutants from contaminated plants and soil. For cattle, separate ingestion rates for grain, forage and silage are available; thus, intake from each of these groups is calculated. Silage is assumed to be derived mainly from vegetative plant parts from a crop such as corn or sorghum. Thus, the plant tissue concentrations are assumed to be similar to those of forage. Unlike forage, however, silage is not consumed by grazing and, therefore, is not assumed to contribute to soil ingestion.

For the remaining animals, a single ingestion rate is available. Sheep are assumed to consume only forage. Hogs are assumed to consume grain and silage. Poultry is assumed to consume only grain. Since hogs and poultry are not grazing animals and no data on soil ingestion is available, hogs and poultry are assumed not to consume soil. Benzo(a)pyrene concentration in animal tissues is calculated according to Equation 5-19.

For beef:

$$\begin{aligned} A_j &= [(0.47\ \text{kg grain} \cdot 1.14\times10^{-8}\ \mu g/g) + (8.8\ \text{kg forage} \cdot 3.06\times10^{-3}\ \mu g/g) + (2.5\ \text{kg silage} \cdot 3.06\times10^{-3}\ \mu g/g) + (0.3\ \text{kg soil} \cdot 7.66\times10^{-5}\ \mu g/g)] \cdot 0.027 \\ &= 9.34\times10^{-4}\ \mu g\ \text{benzo(a)pyrene/g animal tissue} \end{aligned}$$

For beef liver:

$$A_j = [(0.47\ \text{kg grain} \cdot 1.14\text{x}10^{-8}\ \mu g/g) + (8.8\ \text{kg forage} \cdot 3.06\text{x}10^{-3}\ \mu g/g) + (2.5\ \text{kg silage} \cdot 3.06\text{x}10^{-3}\ \mu g/g) + (0.3\ \text{kg soil} \cdot 7.66\text{x}10^{-5}\ \mu g/g)] \cdot 0.036$$

$$= 1.25\text{x}10^{-3}\ \mu g\ \text{benzo(a)pyrene/g animal tissue}$$

For dairy:

$$A_j = [(2.6\ \text{kg grain} \cdot 1.14\text{x}10^{-8}\ \mu g/g) + (11.0\ \text{kg forage} \cdot 3.06\text{x}10^{-3}\ \mu g/g) + (3.3\ \text{kg silage} \cdot 3.06\text{x}10^{-3}\ \mu g/g) + (0.4\ \text{kg soil} \cdot 7.66\text{x}10^{-5}\ \mu g/g)] \cdot 0.004$$

$$= 1.75\text{x}10^{-4}\ \mu g\ \text{benzo(a)pyrene/g animal tissue}$$

For pork:

$$A_j = [(3.0\ \text{kg grain} \cdot 1.14\text{x}10^{-8}\ \mu g/g) + (1.3\ \text{kg silage} \cdot 3.06\text{x}10^{-3}\ \mu g/g)] \cdot 0.019$$

$$= 7.56\text{x}10^{-5}\ \mu g\ \text{B(a)P/g animal tissue}$$

For poultry:

$$A_j = (0.08\ \text{kg grain} \cdot 1.14\text{x}10^{-8}\ \mu g/g) \cdot 0.031$$

$$= 2.83\text{x}10^{-11}\ \mu g\ \text{B(a)P/g animal tissue}$$

For eggs:

$$A_j = (0.08\ \text{kg grain} \cdot 1.14\text{x}10^{-8}\ \mu g/g) \cdot 0.049$$

$$= 4.47\text{x}10^{-11}\ \mu g\ \text{B(a)P/g animal tissue}$$

For lamb:

$$A_j = [(1.4\ \text{kg forage} \cdot 3.06\text{x}10^{-3}\ \mu g/g) + (0.4\ \text{kg soil} \cdot 7.66\text{x}10^{-5}\ \mu g/g)] \cdot 0.020$$

$$= 8.63\text{x}10^{-5}\ \mu g\ \text{B(a)P/g animal tissue}$$

5.5.1.3. CALCULATING HUMAN DAILY INTAKE -- Human daily intake to plants and animals is based on the pollutant concentration in the plant or animal tissue, the consumption rate of the tissue, and the fraction of the food assumed to be produced at home. Because Scenario B is represented by a child who grows up and remains in the area as an adult, this section presents daily intake of both a child and an adult. The adjustments to exposure that are necessary for calculating overall risk are presented in Chapter 15.

The daily intake of benzo(a)pyrene from plants is calculated according Equations 5-22 and 5-23.

Child's intake from grains:

$$Ip_i = 1.14\text{x}10^{-8}\ \mu\text{g B(a)P/g grain} \cdot 0.344 \cdot 5.0\ \text{g DW/kg BW/day}$$
$$= 1.96\text{x}10^{-8}\ \mu\text{g B(a)P/kg BW/day}$$

Adult's intake from grain:

$$Ip_i = 1.14\text{x}10^{-8}\ \mu\text{g B(a)P/g grain} \cdot 0.344 \cdot 1.75\ \text{g DW/kg BW/day}$$
$$= 6.86\text{x}10^{-9}\ \mu\text{g B(a)P/kg BW/day}$$

Child's intake from legumes:

$$Ip_i = 3.06\text{x}10^{-3}\ \mu\text{g B(a)P/g legume} \cdot 0.623 \cdot 1.3\ \text{g DW/kg BW/day}$$
$$= 2.48\text{x}10^{-3}\ \mu\text{g B(a)P/kg BW/day}$$

Adult's intake from legumes:

$$Ip_i = 3.06\text{x}10^{-3}\ \mu\text{g B(a)P/g legume} \cdot 0.623 \cdot 0.53\ \text{g DW/kg BW/day}$$
$$= 1.01\text{x}10^{-3}\ \mu\text{g B(a)P/kg BW/day}$$

Child's intake from potatoes:

$$Ip_i = 1.94\text{x}10^{-6}\ \mu\text{g B(a)P/g potato} \cdot 0.11 \cdot 0.65\ \text{g DW/kg BW/day}$$
$$= 1.39\text{x}10^{-7}\ \mu\text{g B(a)P/kg BW/day}$$

Adult's intake from potatoes:

$$Ip_i = 1.94\text{x}10^{-6}\ \mu\text{g B(a)P/g potato} \cdot 0.11 \cdot 0.285\ \text{g DW/kg BW/day}$$
$$= 6.08\text{x}10^{-8}\ \mu\text{g B(a)P/kg BW/day}$$

Child's intake from root vegetables:

Ip_i = 1.94×10^{-6} μg B(a)P/g root · 0.347 · 0.07 g DW/kg BW/day

= 4.71×10^{-8} μg B(a)P/kg BW/day

Adult's intake from root vegetables:

Ip_i = 1.94×10^{-6} μg B(a)P/g root · 0.347 · 0.03 g DW/kg BW/day

= 2.02×10^{-8} μg B(a)P/kg BW/day

Child's intake from fruits:

Ip_i = 3.06×10^{-3} μg B(a)P/g fruit · 0.212 · 2.08 g DW/kg BW/day

= 1.35×10^{-3} μg B(a)P/kg BW/day

Adult's intake from fruits:

Ip_i = 3.06×10^{-3} μg B(a)P/g fruit · 0.212 · 0.33 g DW/kg BW/day

= 2.14×10^{-4} μg B(a)P/kg BW/day

Child's intake from fruiting vegetables:

Ip_i = 3.06×10^{-3} μg B(a)P/g fruit · 0.308 · 0.24 g DW/kg BW/day

= 2.26×10^{-4} μg B(a)P/kg BW/day

Adult's intake from fruiting vegetables:

Ip_i = 3.06×10^{-3} μg B(a)P/g fruit · 0.308 · 0.105 g DW/kg BW/day

= 9.90×10^{-5} μg B(a)P/kg BW/day

Child's intake from leafy vegetables:

Ip_i = 3.06×10^{-3} μg B(a)P/g leafy · 0.178 · 0.02 g DW/kg BW/day

= 1.09×10^{-5} μg B(a)P/kg BW/day

Adult's intake from leafy vegetables:

Ip_i = 3.06×10^{-3} μg B(a)P/g leafy · 0.178 · 0.02 g DW/kg BW/day

= 1.09×10^{-5} μg B(a)P/kg BW/day

The total daily intake from plants is the sum of the benzo(a)pyrene concentration of each plant group:

Total Child's Intake from Plants:

$$Ip = (1.96\text{x}10^{-8} + 2.48\text{x}10^{-3} + 1.39\text{x}10^{-7} + 4.71\text{x}10^{-8} + 1.35\text{x}10^{-3} + 2.26\text{x}10^{-4} + 1.09\text{x}10^{-5}\ \mu g\ B(a)P/kg\ BW/day) \cdot 1\ mg/10^{3}\ \mu g$$

$$\mathbf{= 4.07\text{x}10^{-6}\ mg/kg\ BW/day}$$

Total Adult's Intake from Plants:

$$Ip = (6.86\text{x}10^{-9} + 1.01\text{x}10^{-3} + 6.08\text{x}10^{-8} + 2.02\text{x}10^{-8} + 2.14\text{x}10^{-4} + 9.90\text{x}10^{-5} + 1.09\text{x}10^{-5}\ \mu g\ B(a)P/kg\ BW/day) \cdot 1\ mg/10^{3}\ \mu g$$

$$\mathbf{= 1.33\text{x}10^{-6}\ mg/kg\ BW/day}$$

The daily intake from animal foods is calculated according to Equations 5-23 and 5-24:

Child's intake from beef:

$$Ia_j = 9.34\text{x}10^{-4}\ \mu g\ B(a)P/g\ beef \cdot 0.039 \cdot 1.713\ g\ beef/kg\ BW/day$$

$$= 6.24\text{x}10^{-5}\ \mu g\ B(a)P/kg\ BW/day$$

Adult's intake from beef:

$$Ia_j = 9.34\text{x}10^{-4}\ \mu g\ B(a)P/g\ beef \cdot 0.039 \cdot 0.555\ g\ beef/kg\ BW/day$$

$$= 2.02\text{x}10^{-5}\ \mu g\ B(a)P/kg\ BW/day$$

Child's intake from beef liver:

$$Ia_j = 1.25\text{x}10^{-3}\ \mu g\ B(a)P/g\ liver \cdot 0.020 \cdot 0.032\ g\ liver/kg\ BW/day$$

$$= 8.00\text{x}10^{-7}\ \mu g\ B(a)P/kg\ BW/day$$

Adult's intake from beef liver:

Ia_j = 1.25×10^{-3} μg B(a)P/g liver · 0.020 · 0.016 g liver/kg BW/day

= 4.00×10^{-7} μg B(a)P/kg BW/day

Child's intake from pork:

Ia_j = 7.56×10^{-5} μg B(a)P/g pork · 0.034 · 1.325 g pork/kg BW/day

= 3.41×10^{-6} μg B(a)P/kg BW/day

Adult's intake from pork:

Ia_j = 7.56×10^{-5} μg B(a)P/g pork · 0.034 · 0.338 g pork/kg BW/day

= 8.69×10^{-7} μg B(a)P/kg BW/day

Child's intake from poultry:

Ia_j = 2.83×10^{-11} μg B(a)P/g poultry · 0.036 · 0.487 g poultry/kg BW/day

= 4.96×10^{-13} μg B(a)P/kg BW/day

Adult's intake from poultry:

Ia_j = 2.83×10^{-11} μg B(a)P/g poultry · 0.036 · 0.115 g poultry/kg BW/day

= 1.17×10^{-13} μg B(a)P/kg BW/day

Child's intake from eggs:

Ia_j = 4.47×10^{-11} μg B(a)P/g eggs · 0.022 · 0.734 g eggs/kg BW/day

= 7.23×10^{-13} μg B(a)P/kg BW/day

Adult's intake from eggs:

Ia_j = 4.47×10^{-11} μg B(a)P/g eggs · 0.022 · 0.123 g eggs/kg BW/day

= 1.21×10^{-13} μg B(a)P/kg BW/day

Child's intake from lamb:

Ia_j = $8.63\text{x}10^{-5}$ μg B(a)P/g lamb · 0.020 · 0.017 g lamb/kg BW/day

= $2.93\text{x}10^{-8}$ μg B(a)P/kg BW/day

Adult's intake from lamb:

Ia_j = $8.63\text{x}10^{-5}$ μg B(a)P/g lamb · 0.020 · 0.006 g lamb/kg BW/day

= $1.04\text{x}10^{-8}$ μg B(a)P/kg BW/day

Dairy products from two sources are assumed to contribute to exposure: a portion produced at home, within a 5 km area of the combustor, and the remainder produced commercially within a 50 km area of the combustor. Thus, determining exposure from dairy products requires two calculations. The pollutant concentration in dairy products (A_j dairy) originating from 50 km was calculated as shown on page 5-45 using the values of soil concentration and deposition appropriate for the 50 km area.

Child's intake from dairy:

Ia_j (5km) = $1.75\text{x}10^{-4}$ μg B(a)P/g dairy · 0.003 · 3.975 g dairy/ kg BW/day

= $2.09\text{x}10^{-6}$ μg B(a)P/kg BW/day

Ia_j (50km) = $8.05\text{x}10^{-3}$ μg B(a)P/g dairy · 0.997 · 3.975 g dairy/ kg BW/day

= $3.19\text{x}10^{-2}$ μg B(a)P/kg BW/day

Total Ia_j = $2.09\text{x}10^{-6}$ + $3.19\text{x}10^{-2}$ μg B(a)P/kg BW/day

= $3.19\text{x}10^{-2}$ μg B(a)P/kg BW/day

Adult's intake from dairy:

$$Ia_j \text{ (5km)} = 1.75\text{x}10^{-4}\ \mu g\ B(a)P/g\ dairy \cdot 0.003 \cdot 0.733\ g\ dairy/kg\ BW/day$$

$$= 3.85\text{x}10^{-7}\ \mu g\ B(a)P/kg\ BW/day$$

$$Ia_j \text{ (50km)} = 8.05\text{x}10^{-3}\ \mu g\ B(a)P/g\ dairy \cdot 0.997 \cdot 0.733\ g\ dairy/kg\ BW/day$$

$$= 5.88\text{x}10^{-3}\ \mu g\ B(a)P/kg\ BW/day$$

$$\text{Total } Ia_j = 3.85\text{x}10^{-7} + 5.88\text{x}10^{-3}\ \mu g\ B(a)P/kg\ BW/day$$

$$= 5.88\text{x}10^{-3}\ \mu g\ B(a)P/kg\ BW/day$$

The total daily intake from animals is the sum of the benzo(a)pyrene concentration of each meat or animal products group:

Total Child's Intake from Animal Foods:

$$Ia = (6.24\text{x}10^{-5} + 8.00\text{x}10^{-7} + 3.41\text{x}10^{-6} + 4.96\text{x}10^{-13} + 7.23\text{x}10^{-13} + 2.93\text{x}10^{-8} + 3.19\text{x}10^{-2}\ \mu g\ B(a)P/kg\ BW/day) \cdot 1\ mg/10^{3}\ \mu g$$

$$= \mathbf{3.20x10^{-5}\ mg\ B(a)P/kg\ BW/day}$$

Total Adult's Intake:

$$Ia = (2.02\text{x}10^{-5}\ \mu g + 4.00\text{x}10^{-7} + 8.69\text{x}10^{-7} + 1.17\text{x}10^{-13} + 1.21\text{x}10^{-13} + 1.04\text{x}10^{-8} + 5.88\text{x}10^{-3}\ \mu g\ B(a)P/kg\ BW/day) \cdot 1\ mg/10^{3}\ \mu g$$

$$= \mathbf{5.90x10^{-6}\ mg\ B(a)P/kg\ BW/day}$$

5.5.2. Cadmium. This section presents examples of calculating cadmium concentration in plants and animals and calculating total daily intake of cadmium by humans. The input variables used for calculating exposure to cadmium are presented in Table 5-11.

TABLE 5-11

Input Variables For Scenario B Exposure Calculations: Cadmium

Input Variable	Value	Document Section
Sc (μg/g)		4.4.1.
5 km: 20 cm	$6.88x10^{-2}$	
1 cm	$4.06x10^{-1}$	
50 km: 20 cm	$2.87x10^{-3}$	
1 cm	$2.78x10^{-2}$	
Br$_i$ [(μg/g plant DW)/(μg/g soil)]		5.2.1.2.1.[a]
grains	0.05	
legumes	0.24	
potatoes	0.09	
root vegetables	1.98	
fruits	1.16	
fruiting vegetables	1.16	
leafy vegetables	1.18	
forage	0.39	
Dyd ($g/m^2/yr$)		3.6.
5 km	$1.59x10^{-4}$	
50 km	$2.34x10^{-5}$	
Dyw ($g/m^2/yr$)		3.6.
5 km	$2.17x10^{-4}$	
50 km	$6.60x10^{-6}$	
Fw (unitless)	0.02	5.2.2.2.[b]
Rp$_i$ (unitless)		5.2.2.3.[c]
grains	0.0	
legumes	0.008	
potatoes	0.0	
root vegetables	0.0	
fruits	0.05	
fruiting vegetables	0.05	
leafy vegetables	0.16	
forage	0.47	
kp (yr^{-1})	31.88	5.2.2.4.[d]
Tp$_i$ (yr)		5.2.2.5.
forage	0.123	
other plants	0.164	

TABLE 5-11 (cont.)

Input Variable	Value	Document Section
Yp_i (kg DW/m^2)		5.2.2.6.[e]
grains	0.22	
legumes	0.43	
potatoes	0.43	
root vegetables	0.43	
fruits	0.26	
fruiting vegetables	0.26	
leafy vegetables	0.14	
forage	0.31	
Bv_i [(μg/g plant DW)/(μg/g air)]		5.2.3.3.
grains	0.0	
legumes	0.0	
potatoes	0.0	
root vegetables	0.0	
fruits	0.0	
fruiting vegetables	0.0	
leafy vegetables	0.0	
forage	0.0	
Cy (μg/m^3)	0.0	3.6.
Cv (μg/m^3)	0.0	5.2.3.4.
Qp_{ij} (kg DW/day)		5.3.1.
beef-grain	0.47	
beef-forage	8.8	
beef-silage	2.5	
beef liver-grain	0.47	
beef liver-forage	8.8	
beef liver-silage	2.5	
dairy-grain	2.6	
dairy-forage	11.0	
dairy-silage	3.3	
pork-grain	3.0	
pork-silage	1.3	
poultry	0.08	
eggs	0.08	
lamb	1.4	

TABLE 5-11 (cont.)

Input Variable	Value	Document Section
Qs_j (kg DW/day)		5.3.2.
beef	0.3	
beef liver	0.3	
dairy	0.4	
pork	0	
poultry	0	
eggs	0	
lamb	0.4	
Ba_j (d/kg)		5.3.3.[f]
beef	0.003	
beef liver	0.135	
dairy	0.0055	
pork	0.028	
poultry	12.45	
eggs	0.6496	
lamb	0.027	
Fp_i (unitless)		5.4.1.2.[g]
grains	0.344	
legumes	0.623	
potatoes	0.110	
root vegetables	0.347	
fruits	0.212	
fruiting vegetables	0.308	
leafy vegetables	0.178	
Fa_j (unitless)		5.4.2.2.[h]
beef	0.039	
beef liver	0.020	
dairy	0.003	
pork	0.034	
poultry	0.036	
eggs	0.022	
lamb	0.020	

TABLE 5-11 (cont.)

Input Variable	Value		Document Section
Cp_i (g DW/kg BW/day)	Children	Adults[i]	5.4.1.1.
grains	5.0	1.75	
legumes	1.3	0.530	
potatoes	0.65	0.285	
root vegetables	0.07	0.03	
fruits	2.08	0.33	
fruiting vegetables	0.24	0.105	
leafy vegetables	0.02	0.02	
Ca_j (g DW/kg BW/day)	Children	Adults	5.4.2.1.
beef	1.713	0.555	
beef liver	0.032	0.016	
dairy	3.975	0.733	
pork	1.325	0.338	
poultry	0.487	0.115	
eggs	0.734	0.123	
lamb	0.016	0.006	

---

[a] Sources: Risk Assessment of the Dickerson Site (1988) and U.S. EPA (1985b) as cited in Belcher and Travis (1989)

[b] Source: Risk Assessment of the Dickerson Site (1988) as cited in Belcher (1989)

[c] Sources: Baes et al. (1984), Hoffman and Baes (1979) and Shor et al. (1982) as cited in Belcher and Travis (1989)

[d] Sources: Baes et al. (1984) and U.S. EPA (1985b) as cited in Belcher and Travis (1989)

[e] Source: For forage only, Hoffman and Baes (1979) as cited in Belcher and Travis (1989). All other plant groups, Shor et al. (1982) data for Hillsboro County, Florida. Adjusted to DW using water content data in Baes et al. (1984)

[f] Source: Risk Assessment of the Dickerson Site (1988) as cited in Belcher and Travis (1989)

[g] Table 5-7, average of individual foods within each plant group for suburban households

[h] Table 5-9

[i] Table 5-6, average of 70-75th percentile of males and females >13 years old

5.5.2.1. CALCULATING CADMIUM CONCENTRATION IN PLANTS -- Exposure through the following eight plant groups is illustrated: forage, grain, potatoes, leafy vegetables (includes brassica), root vegetables, fruit, fruiting vegetables and legumes. Cadmium concentration in these plants is the sum of concentration due to root uptake and direct deposition of particles onto exposed surfaces. Because cadmium is not volatile, plant concentration due to air-to-plant transfer is not considered. Cadmium concentration in plants due to root uptake (Pr) is calculated according to Equation 5-2.

For grains:

$$\begin{aligned} Pr &= 6.88\times10^{-2}\ \mu g/g \cdot 0.05 \\ &= 3.44\times10^{-3}\ \mu g\ Cd/g\ \text{plant tissue} \end{aligned}$$

For legumes:

$$\begin{aligned} Pr &= 6.88\times10^{-2}\ \mu g/g \cdot 0.24 \\ &= 1.65\times10^{-2}\ \mu g\ Cd/g\ \text{plant tissue} \end{aligned}$$

For potatoes:

$$\begin{aligned} Pr &= 6.88\times10^{-2}\ \mu g/g \cdot 0.09 \\ &= 6.19\times10^{-3}\ \mu g\ Cd/g\ \text{plant tissue} \end{aligned}$$

For root vegetables:

$$\begin{aligned} Pr &= 6.88\times10^{-2}\ \mu g/g \cdot 1.98 \\ &= 1.36\times10^{-1}\ \mu g\ Cd/g\ \text{plant tissue} \end{aligned}$$

For fruits and fruiting vegetables:

$$\begin{aligned} Pr &= 6.88\times10^{-2}\ \mu g/g \cdot 1.16 \\ &= 7.98\times10^{-2}\ \mu g\ Cd/g\ \text{plant tissue} \end{aligned}$$

For leafy vegetables:

$$\begin{aligned} Pr &= 6.88\times10^{-2}\ \mu g/g \cdot 1.18 \\ &= 8.12\times10^{-2}\ \mu g\ Cd/g\ \text{plant tissue} \end{aligned}$$

For forage:

$$Pr = 6.88\text{x}10^{-2}\ \mu g/g \cdot 0.39$$
$$= 2.68\text{x}10^{-2}\ \mu g\ Cd/g \text{ plant tissue}$$

Cadmium concentration in plants due to direct deposition (Pd) is calculated according to Equation 5-4. Pd is not calculated for grains, potatoes, and root vegetables because they are "protected produce" and are not exposed to direct deposition.

For legumes:

$$Pd = \frac{1000 \cdot [1.59\text{x}10^{-4} + (0.02 \cdot 2.17\text{x}10^{-4})] \cdot 0.008 \cdot [1-\exp(-31.88 \cdot 0.164)]}{(0.43)\ (31.88)}$$

$$= 9.51\text{x}10^{-5}\ \mu g\ Cd/g \text{ plant tissue}$$

For fruits and fruiting vegetables:

$$Pd = \frac{1000 \cdot [1.59\text{x}10^{-4} + (0.02 \cdot 2.17\text{x}10^{-4})] \cdot 0.05 \cdot [1-\exp(-31.88 \cdot 0.164)]}{(0.26)\ (31.88)}$$

$$= 9.83\text{x}10^{-4}\ \mu g\ Cd/g \text{ plant tissue}$$

For leafy vegetables:

$$Pd = \frac{1000 \cdot [1.59\text{x}10^{-4} + (0.02 \cdot 2.17\text{x}10^{-4})] \cdot 0.16 \cdot [1-\exp(-31.88 \cdot 0.164)]}{(0.14)\ (31.88)}$$

$$= 5.84\text{x}10^{-3}\ \mu g\ Cd/g \text{ plant tissue}$$

For forage:

$$Pd = \frac{1000 \cdot [1.59\text{x}10^{-4} + (0.02 \cdot 2.17\text{x}10^{-4})] \cdot 0.47 \cdot [1-\exp(-31.88 \cdot 0.123)]}{(0.31)\ (31.88)}$$

$$= 7.75\text{x}10^{-3}\ \mu g\ Cd/g \text{ plant tissue}$$

Cadmium concentration due to air-to-plant transfer (Pv) is not calculated because cadmium is not present in the vapor phase. Therefore, the total cadmium concentration in plants (P_i) is calculated by adding the pollutant concentrations due to root uptake and direct deposition (Equation 5-1).

For grains:

$$P_i = 3.44 \times 10^{-3} \ \mu g\ Cd/g\ tissue$$

For legumes:

$$\begin{aligned} P_i &= 1.65 \times 10^{-2} + 9.51 \times 10^{-5} \\ &= 1.66 \times 10^{-2} \ \mu g\ Cd/g\ tissue \end{aligned}$$

For potatoes:

$$P_i = 6.19 \times 10^{-3} \ \mu g\ Cd/g\ tissue$$

For root vegetables:

$$P_i = 1.36 \times 10^{-1} \ \mu g\ Cd/g\ tissue$$

For fruits and fruiting vegetables:

$$\begin{aligned} P_i &= 7.98 \times 10^{-2} + 9.83 \times 10^{-4} \\ &= 8.08 \times 10^{-2} \ \mu g\ Cd/g\ tissue \end{aligned}$$

For leafy vegetables:

$$\begin{aligned} P_i &= 8.12 \times 10^{-2} + 5.84 \times 10^{-3} \\ &= 8.70 \times 10^{-2} \ \mu g\ Cd/g\ tissue \end{aligned}$$

For forage:

$$\begin{aligned} P_i &= 2.68 \times 10^{-2} + 7.75 \times 10^{-3} \\ &= 3.46 \times 10^{-2} \ \mu g\ Cd/g\ tissue \end{aligned}$$

5.5.2.2. CALCULATING CADMIUM CONCENTRATION IN ANIMAL TISSUES -- Exposure through the following seven animal food groups is illustrated: beef, beef liver, dairy, pork, lamb, poultry, and eggs. Animals ingest pollutants from contaminated plants and soil. For cattle, separate ingestion rates of grain, forage and silage are available; thus, intake from each of these groups is calculated. Silage is assumed to be derived mainly from vegetative plant parts from a crop such as corn or sorghum. Thus, the plant tissue concentrations are assumed to be similar to those of forage. Unlike forage, however, silage is not consumed by grazing and, therefore, is not assumed to contribute to soil ingestion.

For the remaining animals, a single ingestion rate is available. Sheep are assumed to consume only forage. Hogs are assumed to consume grain and silage. Poultry is assumed to consume only grain. Since hogs and poultry are not grazing animals and no data on possible soil ingestion is available, hogs and poultry are assumed not to consume soil. Cadmium concentration in animal tissues is calculated according to Equation 5-19.

For beef:

$$A_i = [(0.47 \text{ kg grain} \cdot 3.44\text{x}10^{-3} \ \mu g/g) + (8.8 \text{ kg forage} \cdot 3.46\text{x}10^{-2} \ \mu g/g) + (2.5 \text{ kg silage} \cdot 3.46\text{x}10^{-2} \ \mu g/g) + (0.3 \text{ kg soil} \cdot 4.06\text{x}10^{-1} \ \mu g/g)] \cdot 0.003$$

$$= 1.54\text{x}10^{-3} \ \mu g \text{ Cd/g animal tissue}$$

For beef liver:

$$A_i = [(0.47 \text{ kg grain} \cdot 3.44\text{x}10^{-3} \ \mu g/g) + (8.8 \text{ kg forage} \cdot 3.46\text{x}10^{-2} \ \mu g/g) + (2.5 \text{ kg silage} \cdot 3.46\text{x}10^{-2} \ \mu g/g) + (0.3 \text{ kg soil} \cdot 4.06\text{x}10^{-1} \ \mu g/g)] \cdot 0.135$$

$$= 6.94\text{x}10^{-2} \ \mu g \text{ Cd/g animal tissue}$$

For dairy:

$$A_i = [(2.6\ \text{kg grain} \cdot 3.44\text{x}10^{-3}\ \mu g/g) + (11.0\ \text{kg forage} \cdot 3.46\text{x}10^{-2}\ \mu g/g) + (3.3\ \text{kg silage} \cdot 3.46\text{x}10^{-2}\ \mu g/g) + (0.4\ \text{kg soil} \cdot 4.06\text{x}10^{-1}\ \mu g/g)] \cdot 0.0055$$

$$= 3.66\text{x}10^{-3}\ \mu g\ \text{Cd/g animal tissue}$$

For pork:

$$A_i = [(3.0\ \text{kg grain} \cdot 3.44\text{x}10^{-3}\ \mu g/g) + (1.3\ \text{kg silage} \cdot 3.46\text{x}10^{-2}\ \mu g/g)] \cdot 0.028$$

$$= 1.55\text{x}10^{-3}\ \mu g\ \text{Cd/g animal tissue}$$

For poultry:

$$A_i = (0.08\ \text{kg grain} \cdot 3.44\text{x}10^{-3}\ \mu g/g) \cdot 12.45$$

$$= 3.43\text{x}10^{-3}\ \mu g\ \text{Cd/g animal tissue}$$

For eggs:

$$A_i = (0.08\ \text{kg grain} \cdot 3.44\text{x}10^{-3}\ \mu g/g) \cdot 0.6496$$

$$= 1.79\text{x}10^{-4}\ \mu g\ \text{Cd/g animal tissue}$$

For lamb:

$$A_i = [(1.4\ \text{kg forage} \cdot 3.46\text{x}10^{-2}\ \mu g/g) + (0.4\ \text{kg soil} \cdot 4.06\text{x}10^{-1}\ \mu g/g)] \cdot 0.027$$

$$= 5.69\text{x}10^{-3}\ \mu g\ \text{Cd/g animal tissue}$$

5.5.2.3. CALCULATING HUMAN DAILY INTAKE -- Human daily intake to plants and animals is based on the pollutant concentration in the plant or animal tissue, the consumption rate of the tissue, and the fraction of the food assumed to be produced at home. Because the B scenario is represented by a child who grows up and remains in the area as an adult, this section presents daily intake of both a child and an adult. The adjustments to exposure that are necessary for calculating overall risk are presented in Chapter 15. The daily intake of cadmium from plants is calculated according Equations 5-22 and 5-23.

Child's intake from grains:

$$
\begin{aligned}
Ip_i &= 3.44\text{x}10^{-3}\ \mu\text{g Cd/g grain} \cdot 0.344 \cdot 5.0\ \text{g DW/kg BW/day} \\
&= 5.92\text{x}10^{-3}\ \mu\text{g Cd/kg BW/day}
\end{aligned}
$$

Adult's intake from grain:

$$
\begin{aligned}
Ip_i &= 3.44\text{x}10^{-3}\ \mu\text{g Cd/g grain} \cdot 0.344 \cdot 1.75\ \text{g DW/kg BW/day} \\
&= 2.07\text{x}10^{-3}\ \mu\text{g Cd/kg BW/day}
\end{aligned}
$$

Child's intake from legumes:

$$
\begin{aligned}
Ip_i &= 1.66\text{x}10^{-2}\ \mu\text{g Cd/g legume} \cdot 0.623 \cdot 1.3\ \text{g DW/kg BW/day} \\
&= 1.34\text{x}10^{-2}\ \mu\text{g Cd/kg BW/day}
\end{aligned}
$$

Adult's intake from legumes:

$$
\begin{aligned}
Ip_i &= 1.66\text{x}10^{-2}\ \mu\text{g Cd/g legume} \cdot 0.623 \cdot 0.530\ \text{g DW/kg BW/day} \\
&= 5.48\text{x}10^{-3}\ \mu\text{g Cd/kg BW/day}
\end{aligned}
$$

Child's intake from potatoes:

$$
\begin{aligned}
Ip_i &= 6.19\text{x}10^{-3}\ \mu\text{g Cd/g potato} \cdot 0.110 \cdot 0.65\ \text{g DW/kg BW/day} \\
&= 4.43\text{x}10^{-4}\ \mu\text{g Cd/kg BW/day}
\end{aligned}
$$

Adult's intake from potatoes:

$$
\begin{aligned}
Ip_i &= 6.19\text{x}10^{-3}\ \mu\text{g Cd/g potato} \cdot 0.110 \cdot 0.285\ \text{g DW/kg BW/day} \\
&= 1.94\text{x}10^{-4}\ \mu\text{g Cd/kg BW/day}
\end{aligned}
$$

Child's intake from root vegetables:

$$
\begin{aligned}
Ip_i &= 1.36\text{x}10^{-1}\ \mu\text{g Cd/g root} \cdot 0.347 \cdot 0.07\ \text{g DW/kg BW/day} \\
&= 3.30\text{x}10^{-3}\ \mu\text{g Cd/kg BW/day}
\end{aligned}
$$

Adult's intake from root vegetables:

$$
\begin{aligned}
Ip_i &= 1.36\text{x}10^{-1}\ \mu\text{g Cd/g root} \cdot 0.347 \cdot 0.03\ \text{g DW/kg BW/day} \\
&= 1.42\text{x}10^{-3}\ \mu\text{g Cd/kg BW/day}
\end{aligned}
$$

Child's intake from fruits:

$$\begin{aligned} Ip_i &= 8.08\text{x}10^{-2}\ \mu\text{g Cd/g fruit} \cdot 0.212 \cdot 2.08\ \text{g DW/kg BW/day} \\ &= 3.56\text{x}10^{-2}\ \mu\text{g Cd/kg BW/day} \end{aligned}$$

Adult's intake from fruits:

$$\begin{aligned} Ip_i &= 8.08\text{x}10^{-2}\ \mu\text{g Cd/g fruit} \cdot 0.212 \cdot 0.33\ \text{g DW/kg BW/day} \\ &= 5.65\text{x}10^{-3}\ \mu\text{g Cd/kg BW/day} \end{aligned}$$

Child's intake from fruiting vegetables:

$$\begin{aligned} Ip_i &= 8.08\text{x}10^{-2}\ \mu\text{g Cd/g fruit} \cdot 0.308 \cdot 0.24\ \text{g DW/kg BW/day} \\ &= 5.97\text{x}10^{-3}\ \mu\text{g Cd/kg BW/day} \end{aligned}$$

Adult's intake from fruiting vegetables:

$$\begin{aligned} Ip_i &= 8.08\text{x}10^{-2}\ \mu\text{g Cd/g fruit} \cdot 0.308 \cdot 0.105\ \text{g DW/kg BW/day} \\ &= 2.61\text{x}10^{-3}\ \mu\text{g Cd/kg BW/day} \end{aligned}$$

Child's intake from leafy vegetables:

$$\begin{aligned} Ip_i &= 8.70\text{x}10^{-2}\ \mu\text{g Cd/g leafy} \cdot 0.178 \cdot 0.02\ \text{g DW/kg BW/day} \\ &= 3.10\text{x}10^{-4}\ \mu\text{g Cd/kg BW/day} \end{aligned}$$

Adult's intake from leafy vegetables:

$$\begin{aligned} Ip_i &= 8.70\text{x}10^{-2}\ \mu\text{g Cd/g leafy} \cdot 0.178 \cdot 0.02\ \text{g DW/kg BW/day} \\ &= 3.10\text{x}10^{-4}\ \mu\text{g Cd/kg BW/day} \end{aligned}$$

The total daily intake from plants is the sum of the cadmium concentration of each plant group:

Total Child's Intake from Plants:

$$\begin{aligned} Ip = {} & (5.92\text{x}10^{-3} + 1.34\text{x}10^{-2} + 4.43\text{x}10^{-4} + 3.30\text{x}10^{-3} + 3.56\text{x}10^{-2} \\ & + 5.97\text{x}10^{-3} + 3.10\text{x}10^{-4}\ \mu\text{g Cd/kg BW/day}) \cdot \quad 1\ \text{mg}/10^{3}\ \mu\text{g} \\ = {} & \mathbf{6.49x10^{-5}\ mg/kg\ BW/day} \end{aligned}$$

Total Adult's Intake from Plants:

$$Ip = (2.07\text{x}10^{-3} + 5.48\text{x}10^{-3} + 1.94\text{x}10^{-4} + 1.42\text{x}10^{-3} + 5.65\text{x}10^{-3} + 2.61\text{x}10^{-3} + 3.10\text{x}10^{-4}\ \mu\text{g Cd/kg BW/day}) \cdot \ 1\ \text{mg}/10^{3}\ \mu\text{g}$$

$$\mathbf{= 1.77\text{x}10^{-5}\ mg/kg\ BW/day}$$

The daily intake from animal foods is calculated according to Equations 5-23 and 5-24:

Child's intake from beef:

$$Ia_j = 1.54\text{x}10^{-3}\ \mu\text{g Cd/g beef} \cdot 0.039 \cdot 1.713\ \text{g beef/kg BW/day}$$
$$= 1.03\text{x}10^{-4}\ \mu\text{g Cd/kg BW/day}$$

Adult's intake from beef:

$$Ia_j = 1.54\text{x}10^{-3}\ \mu\text{g Cd/g beef} \cdot 0.039 \cdot 0.555\ \text{g beef/kg BW/day}$$
$$= 3.33\text{x}10^{-5}\ \mu\text{g Cd/kg BW/day}$$

Child's intake from beef liver:

$$Ia_j = 6.94\text{x}10^{-2}\ \mu\text{g Cd/g liver} \cdot 0.02 \cdot 0.032\ \text{g liver/kg BW/day}$$
$$= 4.44\text{x}10^{-5}\ \mu\text{g Cd/kg BW/day}$$

Adult's intake from beef liver:

$$Ia_j = 6.94\text{x}10^{-2}\ \mu\text{g Cd/g liver} \cdot 0.02 \cdot 0.016\ \text{g liver/kg BW/day}$$
$$= 2.22\text{x}10^{-5}\ \mu\text{g Cd/kg BW/day}$$

Child's intake from pork:

$$Ia_j = 1.55\text{x}10^{-3}\ \mu\text{g Cd/g pork} \cdot 0.034 \cdot 1.325\ \text{g pork/kg BW/day}$$
$$= 6.98\text{x}10^{-5}\ \mu\text{g Cd/kg BW/day}$$

Adult's intake from pork:

$$Ia_j = 1.55\text{x}10^{-3}\ \mu\text{g Cd/g pork} \cdot 0.034 \cdot 0.338\ \text{g pork/kg BW/day}$$
$$= 1.78\text{x}10^{-5}\ \mu\text{g Cd/kg BW/day}$$

Child's intake from poultry:

$$Ia_j = 3.43\text{x}10^{-3}\ \mu\text{g Cd/g poultry} \cdot 0.036 \cdot 0.487\ \text{g poultry/kg BW/day}$$
$$= 6.01\text{x}10^{-5}\ \mu\text{g Cd/kg BW/day}$$

Adult's intake from poultry:

$$Ia_j = 3.43\text{x}10^{-3}\ \mu\text{g Cd/g poultry} \cdot 0.036 \cdot 0.115\ \text{g poultry/kg BW/day}$$
$$= 1.42\text{x}10^{-5}\ \mu\text{g Cd/kg BW/day}$$

Child's intake from eggs:

$$Ia_j = 1.79\text{x}10^{-4}\ \mu\text{g Cd/g eggs} \cdot 0.022 \cdot 0.734\ \text{g eggs/kg BW/day}$$
$$= 2.89\text{x}10^{-6}\ \mu\text{g Cd/kg BW/day}$$

Adult's intake from eggs:

$$Ia_j = 1.79\text{x}10^{-4}\ \mu\text{g Cd/g eggs} \cdot 0.022 \cdot 0.123\ \text{g eggs/kg BW/day}$$
$$= 4.84\text{x}10^{-7}\ \mu\text{g Cd/kg BW/day}$$

Child's intake from lamb:

$$Ia_j = 5.96\text{x}10^{-3}\ \mu\text{g Cd/g lamb} \cdot 0.020 \cdot 0.016\ \text{g lamb/kg BW/day}$$
$$= 1.82\text{x}10^{-6}\ \mu\text{g Cd/kg BW/day}$$

Adult's intake from lamb:

$$Ia_j = 5.96\text{x}10^{-3}\ \mu\text{g Cd/g lamb} \cdot 0.020 \cdot 0.006\ \text{g lamb/kg BW/day}$$
$$= 6.83\text{x}10^{-7}\ \mu\text{g Cd/kg BW/day}$$

Dairy products from two sources are assumed to contribute to exposure: a portion produced at home, within a 5 km area of the combustor, and the remainder produced commercially within a 50 km area of the combustor. Thus, determining exposure from dairy products requires two calculations. The pollutant concentration in dairy products (A_i dairy) originating from 50 km was calculated as shown on page 5-45 using the values of soil concentration and deposition appropriate for the 50 km area.

Child's intake from dairy:

Ia_j (5km) $= 3.66\times10^{-3}\ \mu g$ Cd/g dairy $\cdot$ 0.003 $\cdot$ 3.975 g dairy/kg BW/day
$= 4.36\times10^{-5}\ \mu g$ Cd/kg BW/day

Ia_i (50km) $= 2.39\times10^{-4}\ \mu g$ Cd/g dairy $\cdot$ 0.997 $\cdot$ 3.975 g dairy/kg BW/day
$= 9.47\times10^{-4}\ \mu g$ Cd/kg BW/day

Total Ia_j $= 4.36\times10^{-5} + 9.47\times10^{-4}\ \mu g$ Cd/kg BW/day
$= 9.91\times10^{-4}\ \mu g$ Cd/kg BW/day

Adult's intake from dairy:

Ia_j (5km) $= 3.66\times10^{-3}\ \mu g$ Cd/g dairy $\cdot$ 0.003 $\cdot$ 0.733 g dairy/kg BW/day
$= 8.05\times10^{-6}\ \mu g$ Cd/kg BW/day

Ia_j (50km) $= 2.39\times10^{-4}\ \mu g$ Cd/g dairy $\cdot$ 0.997 $\cdot$ 0.733 g dairy/kg BW/day
$= 1.75\times10^{-4}\ \mu g$ Cd/kg BW/day

Total Ia_j $= 8.05\times10^{-6} + 1.75\times10^{-4}\ \mu g$ Cd/kg BW/day
$= 1.83\times10^{-4}\ \mu g$ Cd/kg BW/day

The total daily intake from animals is the sum of the cadmium concentration of each meat and animal-product group:

Total Child's Intake from Animal Foods:

$$Ia = (1.03\times10^{-4} + 4.44\times10^{-5} + 6.98\times10^{-5} + 6.01\times10^{-5} + 2.89\times10^{-6} + 1.82\times10^{-6} + 9.91\times10^{-4}\ \mu g\ \text{Cd/kg BW/day}) \cdot 1\ \text{mg}/10^{3}\ \mu g$$

$$\mathbf{= 1.27\times10^{-6}\ mg\ Cd/kg\ BW/day}$$

Total Adult's Intake from Animal Foods:

$$Ia = (3.33\times10^{-5} + 2.22\times10^{-5} + 1.78\times10^{-5} + 1.42\times10^{-5} + 4.84\times10^{-7} + 6.83\times10^{-7} + 1.83\times10^{-4}\ \mu g\ \text{Cd/kg BW/day}) \cdot 1\ \text{mg}/10^{3}\ \mu g$$

$$\mathbf{= 2.72\times10^{-7}\ mg\ Cd/kg\ BW/day}$$

5.5.3. Summary. An individual's total food-chain exposure to benzo(a)pyrene and cadmium for Scenario B has been calculated. Exposure as a child and as an adult has been estimated. This exposure is summarized in Table 5-12. The overall risk to the individual will be discussed in Chapter 15.

TABLE 5-12

Summary of Terrestrial Food Chain Exposure for Scenario B

Age/Source of Exposure	Daily Intake (mg/kg BW/day) B(a)P	 Cadmium
Child		
plants	4.07×10^{-6}	6.49×10^{-5}
animals	3.20×10^{-5}	1.27×10^{-6}
total	3.61×10^{-5}	6.62×10^{-5}
Adult		
plants	1.33×10^{-6}	1.77×10^{-5}
animals	5.90×10^{-6}	2.72×10^{-7}
total	7.23×10^{-6}	1.80×10^{-5}

6. DETERMINING EXPOSURE FROM SOIL INGESTION

6.1. INTRODUCTION

Contaminants from combustor emissions deposited on soil can be directly ingested by humans who eat soil intentionally or incidentally by hand-to-mouth transfer. Some children with pica, the intentional ingestion or mouthing of nonfood items, may consume larger quantities of soil. Both adults and children may incidentally ingest small amounts of soil; young children (i.e., preschool, 1-6 years of age) will incidentally consume more soil than adults and older children.

The rate of soil ingestion (i.e., amount/day) and the soil concentration of the contaminant affect exposure by this pathway. Soil ingestion rates for children are based on studies that measure quantities of nonabsorbable tracer minerals in the fecal material of the children. For adults, the soil ingestion rate is determined by making assumptions of the thickness of soil on skin, the number of times hands are placed in the mouth and the exposed surface area of the hands that contacts the mouth.

6.2. HUMAN DAILY INTAKE (DI)

Since this methodology assesses only the risk associated with the increase in exposure due to combustor emissions (incremental risk), the background concentration of contaminants in soil is not included in the quantitation of daily intake.

Human daily intake (DI, mg/kg/day) resulting from ingestion of contaminated soil is a function of soil concentration and soil ingestion rate:

$$DI = Sc \cdot Cs \cdot BW^{-1} \qquad \text{(Equation 6-1)}$$

where:

DI = total human daily intake (mg/kg/day)
Sc = soil concentration of pollutant after the total time of deposition (mg pollutant/g soil)
Cs = soil ingestion rate (g soil/day)
BW = body weight (kg)

The DI for this pathway is compared with the RfD_o (reference dose for chronic oral exposure, mg/kg/day) for systemic toxicants to determine if the contaminant adversely affects human health. If the chemical is a carcinogen, the DI is used with the human cancer potency to determine excess risk (ER). This is further discussed in Chapter 15.

Assumptions and their ramifications of the soil ingestion pathway can be found in Table 6-1.

6.2.1. Soil Concentration (Sc). Humans could possibly be exposed to combustor-emitted contaminants soon after deposition on soil and before contaminants are incorporated into soil layers. Therefore for this pathway, 100% of the deposited contaminant is assumed to be incorporated in the uppermost 1 cm of soil and ingested soil is assumed to originate from the same 1 cm layer.

TABLE 6-1

Assumptions for Soil Ingestion Exposure Pathway

Functional Area	Assumptions	Ramifications/ Limitations
Exposure Assessment	Deposited emissions are not necessarily soil incorporated and may be concentrated in the uppermost soil layer. Deposited contaminant is assumed to be distributed within the uppermost 1 cm of soil, and ingested soil is assumed to originate from the same 1 cm layer.	Overestimates exposure in situations where soil incorporation occurs to any depth >1 cm. If incorporation is to a lesser depth, exposure is underestimated. For example, if exposure is to fallout dust directly, ingested soil could approach 100% deposited particulate.
	Deposition-contaminated soil may be ingested intentionally by children at the rates observed in studies.	Overestimates exposure to the extent the child frequents some areas where deposition has not occurred.
Soil Ingestion Rate	Outdoor soil: Children = 0.1-5 g/day Adults = 0.025-0.05 g/day	May over- or underestimate exposure depending on whether soil and/or dust are considered.

Another depth of soil incorporation may be chosen if it is more appropriate for a site-specific assessment. The soil concentration as determined in Chapter 4.0 should be used as input into Equation 6-1.

6.2.2. Soil Ingestion Rate (Cs). Soil ingestion has been recognized as an important source of exposure to several pollutants. Soil ingestion rates vary depending upon the age of the individual, amount of outdoor/indoor activity, frequency of hand-to-mouth contact and seasonal climate. Few studies have been conducted on soil ingestion rates in adults. LaGoy (1987) estimated an ingestion rate of 0.025 g/day for an average adult and 0.05 g/day if the adult exhibits frequent hand-to-mouth contact (e.g., smokers). Children tend to have higher ingestion rates due to more frequent hand-to-mouth contact; also, some children intentionally eat soil. Soil ingestion rates for children range from 0.1 to 5 g/day, with the high extreme of 5 g/day representing children who intentionally eat soil (pica) (U.S. EPA, 1984a).

Studies aimed at more accurately determining the range of ingestion rates (incidental not intentional) have yielded some data, but are as yet inconclusive. Binder et al. (1985, 1986) used tracers (aluminum, silicon and titanium) and determined an average ingestion rate of soil for children to be 108 mg/day with a range of 4-708 mg/day. Clausing et al. (1987) also used tracer elements to determine an ingestion rate for children of 100 mg/day with a range of 21-362 mg/day based only on soil that was taken from the school playground. From these two studies, the average soil ingestion for children incidentally ingesting soil is estimated to be 0.1-0.2 g/day with 0.2 as a reasonably protective value and 1 g/day as an upper-range (U.S. EPA, 1988a).

Seasonal changes also have a role in determining the amount of ingested soil by affecting the amount of time an individual spends outside. In a tropical or subtropical climate, people tend to spend more time outdoors and the opportunity for contact with the contaminated soil would be greater. In a temperate climate, people spend less time outdoors and would most likely ingest contaminated soils at a lower rate.

Hawley (1985) incorporated indoor dust as an additional variable in determining total ingestion of contaminants in soil. Values of 50 mg/day during the six warm months and 100 mg/day in the remaining six months for young children and 0.56 mg/day for adults were assumed for ingestion of dust. LaGoy (1987) suggests that while children under 1 year usually do not come in direct contact with soil, they can be exposed to contaminated house dust with an average ingestion rate of 50 mg/day (250 mg/day maximum).

This methodology uses soil ingestion rates for outdoor exposure in the example calculations. If indoor dust exposure is important in a site-specific assessment, soil ingestion of both outdoor soil and indoor dust could each be determined and added together (Cs [outdoor] + Cs [indoor]).

6.2.3. Body Weight (BW). For this pathway, a body weight of 70 kg is used for adults. A body weight of 17 kg is used for children ages 1-7 years (Nelson et al., 1969) since soil ingestion rates have been derived from data for children in this age range.

6.3. EXAMPLE CALCULATIONS

This section illustrates exposure to benzo(a)pyrene and cadmium via soil ingestion represented by Scenario B. As described in Chapter 2, Scenario B is represented by a child who grows up in an area of moderate deposition and remains in the area for 30 years. Within this time the individual participates in gardening. The values of the input variables chosen for Scenario B are presented in Table 6-2, and are generally in the middle of the range of values discussed earlier.

6.3.1. Benzo(a)Pyrene. The daily intake of benzo(a)pyrene due to ingestion of soil for children can be calculated according to Equation 6-1:

$$\begin{aligned} DI &= 7.66\text{x}10^{-8} \text{ mg/g} \cdot 1 \text{ g/day} \cdot (1/17 \text{ kg}) \\ &= 4.51\text{x}10^{-9} \text{ mg/kg/day} \end{aligned}$$

The daily intake of benzo(a)pyrene due to ingestion of soil for adults can be calculated also using Equation 6-1:

$$\begin{aligned} DI &= 7.66\text{x}10^{-8} \text{ mg/g} \cdot 0.04 \text{ g/day} \cdot (1/70 \text{ kg}) \\ &= 4.38\text{x}10^{-11} \text{ mg/kg/day} \end{aligned}$$

TABLE 6-2

Input Variables for Soil Ingestion Pathway

Input Variable	Value	
	Children	Adults
Sc (mg/g)		(Z = 1 cm and Tc = 60 yr)
B(a)P	7.66×10^{-8}	7.66×10^{-8}
Cadmium	4.06×10^{-4}	4.06×10^{-4}
Cs (g/day)	1	0.04
BW (kg)	17	70

These values are used in Chapter 15 to determine excess risk for exposure to benzo(a)pyrene in combustor emissions.

6.3.2. Cadmium. The daily intake of cadmium due to the ingestion of soil for children can be calculated using Equation 6-1:

$$DI = 4.06\text{x}10^{-4}\ \text{mg/g} \cdot 1\ \text{g/day} \cdot (1/17\ \text{kg})$$
$$= 2.39\text{x}10^{-5}\ \text{mg/kg/day}$$

The daily intake of cadmium due to the ingestion of soil for adults can be calculated using Equation 6-1:

$$DI = 4.06\text{x}10^{-4}\ \text{mg/g} \cdot 0.04\ \text{g/day} \cdot (1/70\ \text{kg})$$
$$= 2.32\text{x}10^{-7}\ \text{mg/kg/day}$$

These values are compared with the RfD_o according to the approach described in Chapter 15.

7. DETERMINING EXPOSURE FROM DERMAL ABSORPTION VIA SOIL

7.1. INTRODUCTION

The Dermal Exposure Model determines exposure from human skin contact with contaminants in the soil. The issue of dermal absorption of deposited contaminants is very complex. There is a fundamental lack of data regarding absorption of chemicals in soil by human skin. Other factors important for estimating human exposure to contaminants by the dermal route (i.e., contact time, contact amount and surface area) also have many uncertainties. The model described here is offered as a possible approach for estimating human exposure and risk associated with dermal contact of deposited emissions. However, in most if not all cases, the available data will not provide a satisfactory basis for risk calculations.

7.2. DAILY DERMAL INTAKE (DDI)

Since this methodology assesses only the risk associated with the increase in exposure due to combustor emissions (incremental risk), the background concentration of contaminants in soil is not included in estimation of dermal daily intake.

The daily dermal intake (DDI, mg/kg/day) represents the increase above background in daily human dermal intake due to the contaminant from combustor emissions. The DDI is determined as follows:

$$DDI = CT \cdot SA \cdot CA \cdot AF \cdot Sc \cdot BW^{-1} \cdot (10^{-3}/24) \qquad \text{(Equation 7-1)}$$

where:

DDI = human daily dermal intake (mg/kg/day)
CT = contact time (hrs/day)
SA = exposed skin surface area (cm^2)
CA = contact amount (mg/cm^2)
AF = absorption fraction (%/day)
Sc = soil concentration (mg/g)
BW = body weight (kg)
$10^{-3}/24$ = units conversion factor $(10^{-6}\ g \cdot 1\ day \cdot 10^3 g)/(g \cdot 24\ hrs \cdot mg)$

The assumptions and uncertainties associated with each of these input parameters are discussed in the following sections and summarized in Table 7-1.

To assess whether the contaminant poses a risk to human health by this pathway, the DDI may be compared with a dermal RfD (if available) or may be transformed to an equivalent oral DI and compared with the oral RfD for chronic exposure (RfD_o) for systemic toxicants. If the chemical is a carcinogen, the DDI may be used with the dermal human cancer potency (if available) or may be transformed to an equivalent oral DI and used with the human oral cancer potency to determine excess risk (ER). Assessment of risk to humans is discussed in Chapter 15.

7.2.1. Contact Time (CT). Contact time is defined as the duration (hrs/day) of daily contact with contaminants in soil. The duration of daily contact is measured from the time of first contact to the time skin is cleaned. Duration is affected by both age and time spent outdoors vs. indoors in the vicinity of a combustor.

Age affects duration of contact because children generally spend more of their time playing outside and retain soil on their

TABLE 7-1

Assumptions and Uncertainties for the Dermal Exposure Model

Functional Area	Assumptions/ Uncertainty	Ramifications/ Limitations
Exposure Assessment	Deposited emissions are not necessarily soil incorporated and may concentrate in the uppermost soil layer. Deposited contaminant is assumed to be distributed within the uppermost 1 cm of soil, and the soil that contacts the skin is assumed to originate from the same 1 cm layer.	Overestimates exposure in situations where soil incorporation occurs to any depth >1 cm
Dermal intake	Linear with respect to soil concentration	May overestimate exposure
Fraction of contaminant absorbed	Chemical- and matrix-specific	Uncertainty for many contaminants and matrices
	Data needed for dermal absorption in humans may not be available	Uncertainty between species extrapolation for dermal exposure; uncertainty in route-to-route extrapolation of absorption
	Dermal absorption varies with age	Absorption data for a contaminant at one age may over- or underestimate absorption at another age
	May be concentration dependent	Study data at one concentration may over- or underestimate actual absorption at another concentration

TABLE 7-1 (cont.)

Functional Area	Assumption/ Uncertainty	Ramifications/ Limitations
Contact time	Outdoor soil: Children = 4-12 hrs/day Adults = 1-2 hrs/day	May over- or underestimate exposure depending on whether soil and/or dust are considered
Exposed skin	Total intake is proportional to exposed surface area: Adults = 910-2940 cm^2 Children = 300-1400 cm^2	May over- or underestimate exposure
Contact Amount	Outdoor soil: Children = 0.5-1.5 mg/cm^2 Adults = 0.5-3.5 mg/cm^2	May over-or under estimate exposure depending on whether soil and/or dust are considered

skin a for a longer time after coming indoors (U.S. EPA, 1988a). This methodology assumes that children spend from 4-12 hours each day in contact with soil, playing, depending on how much time they spend in school. Adults would spend less time outdoors. This methodology assumes adults spend 1-2 hours each day in contact with soil, working in the yard and garden, after work and on weekends.

Time spent outdoors vs. indoors affects duration of contact with the contaminant because the composition of outdoor soil and indoor dust may be different. Because soil carried into the house contributes to the household dust, about 80% of indoor dust has the same contaminant concentration as outdoor soil. The remaining 20% of indoor dust is particulate matter from cooking, smoking, wearing fabrics, or using sprays and powders (Hawley, 1985).

Hawley (1985) estimated that children would be in contact with outdoor soil for 4-12 hours/day and indoor dust/soil for 16-24 hours/day, depending on the age of the child and the time of year. Adults were estimated to be in contact with outdoor soil for 8 hours/day and indoor dust/soil for 16 hours/day, depending on the season.

The regional climate could increase or decrease the frequency of daily contact with contaminants in soil. In a tropical environment, individuals are likely to spend more time outdoors, increasing the frequency of contact, than in a temperate or arctic environment. Hawley (1985) estimated the contact frequency for children was approximately 5 days/week for 6 months/year, or about 130 days/year, at a duration of 12 hours/day. Adults were estimated to have a contact frequency of approximately 2 days/week for 5 months/year, or about 43 days/year, at a contact duration of 5 hours/day (U.S. EPA, 1988a).

This methodology uses the contact times reported for outdoor soils in the example calculations. If indoor dust is important to a site-specific assessment, contact time to outdoor soil and indoor dust could each be determined and added together. If regional climate is likely to affect the total contact time, a relevant site-specific frequency of contact can be used in conjunction with duration to determine total contact time (CT = hrs/day · days/yr).

7.2.2. Surface Area (SA). Total dermal intake of a contaminant is approximately proportional to the exposed surface area. Depending on the type of clothing worn, exposed surface area may range from 910 - 2940 cm^2 for adults, and 300 - 1400 cm^2 for children (U.S. EPA, 1984b). For example, the exposed surface area of adults wearing short-sleeved, open-necked shirts, pants and shoes, with no gloves or hat, is ≈2940 cm^2, whereas that of children wearing the same clothing is ≈980 cm^2. U.S. EPA (1989a) reports the total mean body surface area for adult males as 19,400 cm^2. The mean surface area for adult males for the hands (1140 cm^2) and forearms (840 cm^2) is 1980 cm^2 or ≈11% of the total body surface area (U.S. EPA, 1989a). The surface area of the forearms comprise approximately one-half the total surface area of the arms. For children aged 2-5 years the 50th percentile for the total body surface area is 5790-7310 cm^2 (males and females; midpoint ≈6550 cm^2)(U.S. EPA, 1989a). The surface area of the hands and forearms for children is, therefore, approximately 750 cm^2 (≈11% of 6550 cm^2).

Regional climate will affect the amount and/or type of clothing worn. Obviously, people in cold climates may have less exposed surface area than people in tropical climates. Risk assessors should consider site-specific conditions when conducting a risk assessment and should choose the high or low end of the range accordingly.

7.2.3. Contact Amount (CA). Contact amount is defined as the amount of soil accumulating on skin and is assumed to be in the range of 0.5 to 1.5 mg/cm^2 for both children and adults. This range also represents an average for the entire exposed area (U.S. EPA, 1984b; U.S. EPA, 1988a based on the reports of Snyder, 1975; Lepow et al., 1975; and Roels et al., 1980). Soil on the skin originates from both outdoor soil or indoor dust, although outdoor soil contributes more significantly to overall dermal exposure (Hawley, 1985). Based on the weight of soil on exposed hands, Hawley (1985) assumed a contact amount of 3.5 mg/cm^2 for adults doing yard work and 0.51 mg/cm^2 for children playing outdoors.

Hawley (1985) also estimated the average dust covering indoors, 560 mg/m^2, based on a range of 110-590 mg/m^2 reported in Solomon and Hartford (1976) and Schaefer et al. (1972). From this estimate, Hawley (1985) calculated contact amount of indoor dust to be 0.56 mg/cm^2 for young children with hands, feet, and forearms exposed and older children with only hands exposed. Contact amount of indoor dust for adults with hands and/or arms exposed was 0.56 - 1.8 mg/cm^2.

This methodology will use the outdoor soil contact amount in the example calculations. In a site-specific assessment, this parameter could be modified to include indoor dust contact (CA= CA[outdoor] + CA[indoor]).

7.2.4. Absorption Fraction (AF). The absorption fraction is the fraction of the applied dose of the compound absorbed by human skin in one day (i.e., 6%/24 hours). If no specific data are available, a conservative assumption may be made. The assumption of complete absorption (i.e., AF=1) would be the most conservative approach, but may be unrealistic. For example, Kimbrough et al. (1984) suggested human dermal absorption of TCDD from soil was 1%. This value is most likely an upper estimate of what is likely to be absorbed by humans (Shu et al., 1988).

Dermal absorption of a contaminant depends on the physicochemical properties of the contaminant and the matrix (U.S. EPA, 1982c; Hawley, 1985) as well as the characteristics of the skin. Properties of the contaminant such as lipid solubility, pH and molecular size will affect dermal absorption. The fraction of the contaminant absorbed may vary with concentration (Feldman and Maibach, 1974). Shu et al. (1988), however, concluded that, in the rat, dermal bioavailability of TCDD in soil was not altered by concentration of the chemical. This methodology conservatively assumes that absorption of the contaminant increases continuously with the contaminant concentration in soil.

Absorption of a contaminant contained in a matrix or vehicle is affected by the physicochemical properties of the matrix, but there is uncertainty as to how these soil or particle types influence absorption of deposited contaminants. Poiger and Schlatter (1980) demonstrated that a soil matrix reduced the absorption of TCDD when TCDD was applied to the skin of rats in a soil-water paste.

Absorption of contaminants may be altered by skin damage (disease, laceration or abrasion), age, anatomical location (U.S. EPA, 1982b; Maibach et al., 1971), temperature and humidity; however, the specific way in which these factors affect the absorption is unclear due to a scarcity of information.

Estimating the absorbed fraction of a contaminant is very complex and depends on many factors as discussed above. A paucity of data regarding the dermal absorption of chemicals in humans, particularly from particulate or soil matrices, will make estimating the fraction of contaminant absorbed from soil difficult for this methodology. In many cases, DDI will not be able to be calculated due to lack of data.

Values of absorption fractions may not be available for use in exposure assessment. One approach might be to extrapolate absorption fractions from other types of studies. Although studies of absorption by the dermal route of exposure may not be available in the literature, studies of absorption by other routes are often available. However, route-to-route extrapolation poses uncertainties, and guidelines for this type of extrapolation have not been clearly outlined. In addition, extrapolation of dermal absorption data among animal species introduces uncertainty to the assessment.

7.2.5. Soil Concentration (Sc). Humans could possibly be exposed to contaminants soon after deposition on soil and before contaminants are incorporated into soil layers. Individuals would then be exposed to the greatest concentrations of the contaminants. Therefore for this pathway, 100% of the deposited contaminant is assumed to be incorporated in the uppermost 1 cm of soil and soil available for contact is assumed to originate from the same 1 cm layer. Another depth of soil incorporation may be chosen if it is more appropriate for a site-specific assessment. The soil concentration as determined in Chapter 4 should be used as input into Equation 7-1.

7.2.6. Body Weight (BW). For this pathway, a body weight of 70 kg is used for adults. A body weight of 16.7 kg is used for children ages 2-5 years (Nelson et al., 1969) since surface area measures are approximated for children in this age range.

7.3. EXAMPLE CALCULATIONS

This section illustrates a dermal absorption to benzo(a)pyrene and cadmium represented by Scenario B. As presented in Chapter 2, Scenario B is represented by a child who grows up in an area where pollutants have been emitted for the 60 years of combustor operation. The child remains in the area for 30 years. The individual gardens and is exposed by dermal contact to pollutants in soil. The values of the input variables chosen for Scenario B are generally in the middle of the range of values discussed earlier and are presented in Table 7-2. Surface area for exposed hands and forearms was used.

7.3.1. Benzo(a)Pyrene. Exposure to benzo(a)pyrene from dermal absorption of soil for children can be calculated according to Equation 7-1:

$$\begin{aligned} \text{DDI} &= 8\ \text{hr/day} \cdot 750\ \text{cm}^2 \cdot 1.0\ \text{mg/cm}^2 \cdot \text{AF} \cdot 7.66\text{x}10^{-8} \\ &\quad \text{mg/g} \cdot (1/16.7\ \text{kg}) \cdot (1\ \text{day} \cdot 10^{-3}\text{g})/(24\ \text{hr} \cdot \text{mg}) \\ &= \text{Cannot be quantitated because input data for AF are not available} \end{aligned}$$

Exposure to benzo(a)pyrene from dermal absorption for adults can be calculated using Equation 4-1:

$$\begin{aligned} \text{DDI} &= 1.5\ \text{hr/day} \cdot 1980\ \text{cm}^2 \cdot 1.0\ \text{mg/cm}^2 \cdot \text{AF} \cdot 7.66\text{x}10^{-8} \\ &\quad \text{mg/g} \cdot (1/70\ \text{kg}) \cdot (1\ \text{day} \cdot 10^{-3}\text{g})/(24\ \text{hr} \cdot \text{mg}) \\ &= \text{Cannot be quantitated because input data for AF are not available} \end{aligned}$$

If data were available, the DDI could be used in Chapter 15 to determine excess risk for exposure to benzo(a)pyrene in combustor emissions.

TABLE 7-2

Input Variables for Dermal Exposure Model

Input Variable	Values Children	 Adults
CT (hrs/day)	8	1.5
SA (cm^2)	750	1980
CA (mg/cm^2)	1.0	1.0
AF (%/day)	NA	NA
Sc (mg/g) (Z = 1 cm and Tc = 60 yr)		
B(a)P	7.66×10^{-8}	7.66×10^{-8}
Cadmium	4.06×10^{-4}	4.06×10^{-4}
BW (kg)	16.7	70

NA: Data not available

7.3.2. Cadmium. Exposure to cadmium from dermal absorption for children can be calculated using Equation 4-1:

$$\text{DDI} = 8\ \text{hr/day} \cdot 750\ \text{cm}^2 \cdot 1.0\ \text{mg/cm}^2 \cdot \text{AF} \cdot 4.06\text{x}10^{-4}\ \text{mg/g} \cdot (1/16.7\ \text{kg}) \cdot (1\ \text{day} \cdot 10^{-3}\text{g})/(24\ \text{hr} \cdot \text{mg})$$

= Cannot be quantitated because input data for AF are not available

Exposure to cadmium from dermal absorption for adults can be calculated using Equation 4-1:

$$\text{DDI} = 1.5\ \text{hr/day} \cdot 1980\ \text{cm}^2 \cdot 1.0\ \text{mg/cm}^2 \cdot \text{AF} \cdot 4.06\text{x}10^{-4}\ \text{mg/g} \cdot (1/70) \cdot (1\ \text{day} \cdot 10^{-3}\text{g})/(24\ \text{hr} \cdot \text{mg})$$

= Cannot be quantitated because input data for AF are not available

If data were available, the DI could be used for comparison to the RfD according to the approach described in Chapter 15.

8. DUST RESUSPENSION

8.1. INTRODUCTION

Pollutants in the soil can be resuspended as dust by wind erosion. A particular fraction of these particles, that is those ≤ 10 μm in diameter, may be inhaled by persons (U.S. EPA, 1988b). Methodologies have been developed to assess human exposure to pollutants resuspended by wind from surface contamination sites. This section describes the process of soil erosion and the factors that influence dust resuspension, and provides two examples to illustrate its use. The calculations were performed using the methodology described in Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites (U.S. EPA, 1985a) to estimate short-term (24-hour) and long-term (annual average) exposure. For both examples, the annual emission rates for dust resuspension are less than the stack emission rate.

8.2. SOIL EROSION PROCESS

The process of soil erosion is a function of continual soil motion: surface creep, saltation and suspension. Surface creep describes the rolling or sliding movement of particles 500-1000 μm in diameter (Sehmel, 1980). These surface creep particles are too heavy to be lifted up by the wind, but roll and exchange momentum during collision with smaller particles, which undergo saltation. Saltation is leaping and bouncing particles lifted by the wind that are too heavy to remain suspended. Saltation particles range from 100 to 500 μm in diameter (Sehmel, 1980). Suspension of particles occurs when saltation particles collide with smaller particles, that is particles <100 μm. If saltation was prevented, the suspension particles would not be readily eroded (Sehmel, 1980).

8.3. FACTORS AFFECTING WIND EROSION

Soil erosion is affected by soil characteristics and wind. The factors affecting dust resuspension include: moisture content, presence of nonerodible surfaces (i.e., rocks and vegetation), surface crust stability, wind velocity, particle size, and soil roughness height.

Moisture hinders the resuspension of dust by forming a cohesive matrix among the dust or soil particles. When the soil is saturated, erosion will not occur (Sehmel, 1980). As the surface layer moisture is removed by evaporation, resuspension occurs. Evaporation is influenced by wind speed, temperature and soil characteristics.

The presence of nonerodible surfaces reduces the amount of dust resuspension by decreasing the soil surface area exposed to the wind. Nonerodible surfaces include clumps of grass, and stones >1 cm in diameter. Soil surfaces containing nonerodible surfaces are considered to be "limited reservoirs" because of finite availability and threshold friction velocities >75 cm/s (U.S. EPA, 1985a). Areas that are highly erodible have threshold friction velocities <75 cm/s and are considered to have unlimited erosion potential (U.S. EPA, 1985a). For example, an area covered with a continuous layer of grass has minimal soil erosion by wind as compared with an agricultural field consisting of erodible particles.

The surface crust, which is composed of small particles, hinders wind erosion and evaporation. Thus, crusted surfaces have greater threshold friction velocities. Soils without crusts such as sandy or disturbed soils are more erodible (U.S. EPA, 1985a).

Particle size influences the wind speed required to initiate dust resuspension by wind erosion. Particles ranging in diameter from 100 to 150 μm have the lowest threshold wind speeds. As the particle size decreases below 100 μm the threshold speed increases. For particles >150 μm, the threshold speed also increases. If particle sizes are mixed, the threshold wind speed is less than the speed required to erode the largest particle (Sehmel, 1980).

Soil roughness height is related to height, width and spacing of clods and furrows. This parameter is required to convert the threshold friction velocity to a wind speed comparable with the wind speeds at 7 m above the surface (U.S. EPA, 1985a). Coward and Guenther (1976) developed the roughness height scale for various conditions of ground cover, such as natural snow and plowed field.

8.4. EXAMPLE CALCULATIONS

Some methodologies have been developed by the U.S. EPA to determine the total wind erosion soil loss resulting from surface creep, saltation, and suspension (Skidmore and Woodruff, 1968; Sehmel, 1980; and Smith et al., 1982) while one other methodology determines the soil erosion loss for only the portion that is both suspendible and respirable (U.S. EPA, 1985a). To assess human exposure to pollutants resuspended by wind erosion (U.S. EPA, 1985a), this latter methodology has been applied. In applying this method to deposited incinerator emissions, dust resuspension yields an emission rate that can be compared with the incinerator stack emission rate. Both of these emission rates are input for air dispersion modeling, which predicts ambient air concentrations to which persons are exposed. Below are two examples that illustrate the emission rate of the resuspended dust is less than or equal to the stack emission rate. The equations used to calculate both the annual average emission rate and the worst-case 24-hr emission rate were taken from Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Site (U.S. EPA, 1985a).

Both examples compare the cadmium emission rate from the stack of an incinerator with the cadmium emission rate from dust resuspended by wind erosion. The stack emission rate of cadmium was based on an incinerator in western Florida and was previously estimated to be 1.9×10^4 µg/s (PEI Associates, Inc. and H.E. Cramer Company, Inc., 1986). This section presents a soil concentration modeled for the Florida incinerator. It also presents an estimate of average annual emission rate and worst-case 24-hr emission rate for two hypothetical sources: a 10,000 m^2 field in western Florida and a 400,000 m^2 field in North Platte, Nebraska.

8.4.1. Estimation of Soil Concentration. The soil concentration of cadmium for each example was estimated by using the deposition rate modeled for the Florida incinerator and assuming the soil type is silt/loam. For Florida, the soil concentration of cadmium is 4.06×10^{-4} mg/g as calculated for Scenario B in Section 4.1.1. In order to calculate the soil concentration for Nebraska, the soil loss constant of cadmium for silt/loam soils is needed. The soil loss constant for losses due to leaching is estimated for each example by using geographic-specific parameters in Equation 4-3.

For North Platte, Nebraska, the soil loss constant due to leaching is calculated by:

$$ksl = \frac{P + I - Ev}{\theta \cdot Z \cdot [1.0 + (BD \cdot K_d/\theta)]}$$

where:

P = 50 cm/yr (Baes et al., 1984)
I = 40 cm/yr (Baes et al., 1984)
Ev = 62.5 cm/yr (Baes et al., 1984)
θ = 0.22 mL/cm^3
Z = 1 cm
BD = 1.5 g/cm^3 (NRC, 1984)
K_d = 148 mL/g

$$ksl = \frac{50\ \text{cm/yr} + 40\ \text{cm/yr} - 62.5\ \text{cm/yr}}{0.22\ \text{mL/cm}^3 \cdot 1.0\ \text{cm} \cdot [1.0 + (1.5\ \text{g/cm}^3 \cdot 148\ \text{mL/g}/0.22\ \text{mg/cm}^3)]}$$

$$= 0.18\ \text{yr}^{-1}$$

The soil loss constant due to leaching (ksl) is assumed to equal the soil loss constant for all processes (ks) since cadmium does not undergo degradation (abiotic and biotic) or volatilization. The cadmium soil concentration is estimated by using Equation 4-1 and by assuming soil depth is 1 cm, total time period for emissions is 60 years, and soil loss constant due to leaching is equal to soil loss constant for all processes. For Nebraska, cadmium soil concentration is calculated by:

$$Sc = \frac{(Dyd + Dyw) \cdot [1.0 - \exp(-ks \cdot Tc)] \cdot 0.1}{Z \cdot BD \cdot ks}$$

where:

Dyd = $1.59\mathrm{x}10^{-4}$ g/m^2/yr
Dyw = $2.17\mathrm{x}10^{-4}$ g/m^2/yr
ks = 0.18 yr^{-1}
Tc = 60 yr
Z = 1 cm
BD = 1.5 g/cm^3

$$Sc = \frac{(1.59\mathrm{x}10^{-4} + 2.17\mathrm{x}10^{-4}) \cdot [1.0 - \exp(-0.18 \cdot 60)] \cdot 0.1}{1 \cdot 1.5 \cdot 0.18}$$

$$= 1.39\mathrm{x}10^{-4} \text{ mg Cd/g soil}$$

The soil concentrations of cadmium due to the deposition of incinerator emissions are $4.06\mathrm{x}10^{-4}$ mg/g for Florida and $1.39\mathrm{x}10^{-4}$ mg/g for Nebraska.

8.4.2. Annual Average Emission Rate. The annual emission rate of contaminant as respirable particles (R_{10}) is the product of the area of the field, the pollutant concentration in the soil and the emission factor of respirable particulate matter, those $\leq 10\ \mu m$ diameter (PM_{10}) (U.S. EPA, 1985a).

$$R_{10} = A \cdot \alpha \cdot E_{10} \qquad \text{(Equation 8-1)}$$

where:

R_{10} = annual emission rate of the contaminant (mg/s)
A = area of the source (m^2)
α = pollutant concentration in the PM_{10} fraction of soil (mg pollutant/mg soil)
E_{10} = annual average PM_{10} emission factor ($g/m^2/hr$)

For both examples, the contaminated site is assumed to be unvegetated with no nonerodible elements and no evidence of crusting (U.S. EPA, 1985a). Since both examples are hypothetical, their wind erosion threshold friction velocities are assumed to be 50 cm/s (U.S. EPA, 1985a), and their roughness heights are assumed to be 2.0 cm (for grasslands) (U.S. EPA, 1985a). Threshold friction velocity is usually determined by hand sieving during a site investigation; field procedures are outlined in U.S. EPA (1985a). For the first example, the hypothetical source is a field in Tampa, Florida with an area, A, of 10,000 m^2. For the second example, a field in North Platte, Nebraska, the area, A, is assumed to be 400,000 m^2. (Both examples are modifications of an example in Rapid Assessment of Exposure to Particulate Emissions from

Surface Contamination Sites (U.S. EPA, 1985a) with various parameter values changed to represent the conditions of fields contaminated by combustor emissions.)

Calculation of E_{10} and R_{10} for the two examples are described below. Choosing the appropriate calculation for E_{10} depends on whether the field in question is a "limited" or "unlimited" reservoir. Fields with wind erosion threshold velocity of <75 cm/s are considered "unlimited reservoirs". For both examples the threshold erosion friction velocity is <75 cm/s, so both fields are categorized as "unlimited reservoirs". The annual average emission rate of contaminated surface with unlimited erosion potential (E_{10}) is calculated by:

$$E_{10} = 0.036 \cdot (1-V) \cdot ([u]/u_t)^3 \cdot F(x) \quad \text{(Equation 8-2)}$$

where:

E_{10} = emission factor of respirable (PM_{10}) particles ($g/m^2/hr$)
V = fraction of contaminated surface vegetation cover
$[u]$ = mean annual wind speed (m/s)
u_t = threshold wind speed (m/s)
x = $0.886\ u_t/[u]$
$F(x)$ = function of x (derived from Figure 8-1)

For the first example, the field is assumed to be bare, so V is assumed to be 0. The mean annual wind speed for Tampa is 3.9 m/s (U.S. EPA, 1985a) and is used to calculate threshold wind speed. Threshold wind speed is calculated by:

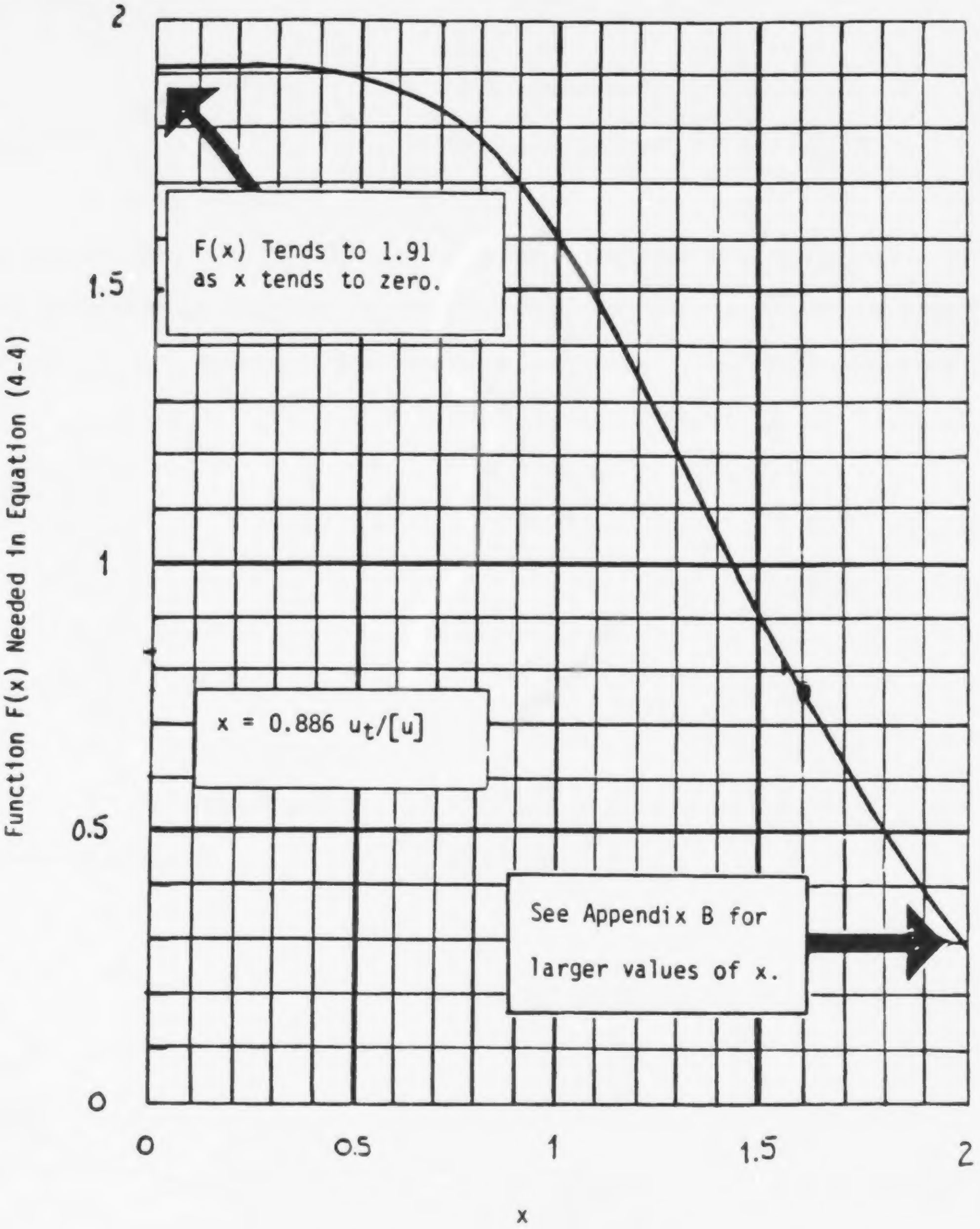


Figure 8-1. Graph of Function F(x) Needed to Estimate Unlimited Erosion

Source: U.S. EPA, 1985a

$$u_t = R_w \cdot F_v \qquad \text{(Equation 8-3)}$$

where:

u_t = threshold wind speed (m/s)
R_w = ratio of wind speed at 7m to friction velocity as a function of roughness height
F_v = wind erosion threshold friction velocity (m/s)

For the Florida example, the wind erosion friction velocity is assumed to be 0.50 m/s and the roughness height is assumed to be 2.0 cm for grassland. Rw for a roughness height of 2.0, derived from Figure 8-2, is 15.0.

$$u_t = R_w \cdot F_v$$

where:

R_w = 15.0
F_v = 0.50 m/s

$$u_t = 15.0 \cdot 0.50 \text{ m/s}$$

$$= 7.5 \text{ m/s}$$

Therefore, x can be calculated by the following equation.

$$x = 0.886 \cdot (u_t/[u]) \qquad \text{(Equation 8-4)}$$

where:

u_t = 7.5 m/s
$[u]$ = 3.9 m/s (U.S. EPA, 1985a)

$$x = 0.886 \cdot (7.5 \text{ m/s}/3.9 \text{ m/s})$$

$$= 1.7$$

Using Figure 8-1, F(x) = 0.62.

E_{10} for the field in western Florida is calculated by using equation 8-2.

$$E_{10} = 0.036\ (1-V)\ ([u]/u_t)^3\ F(x)$$

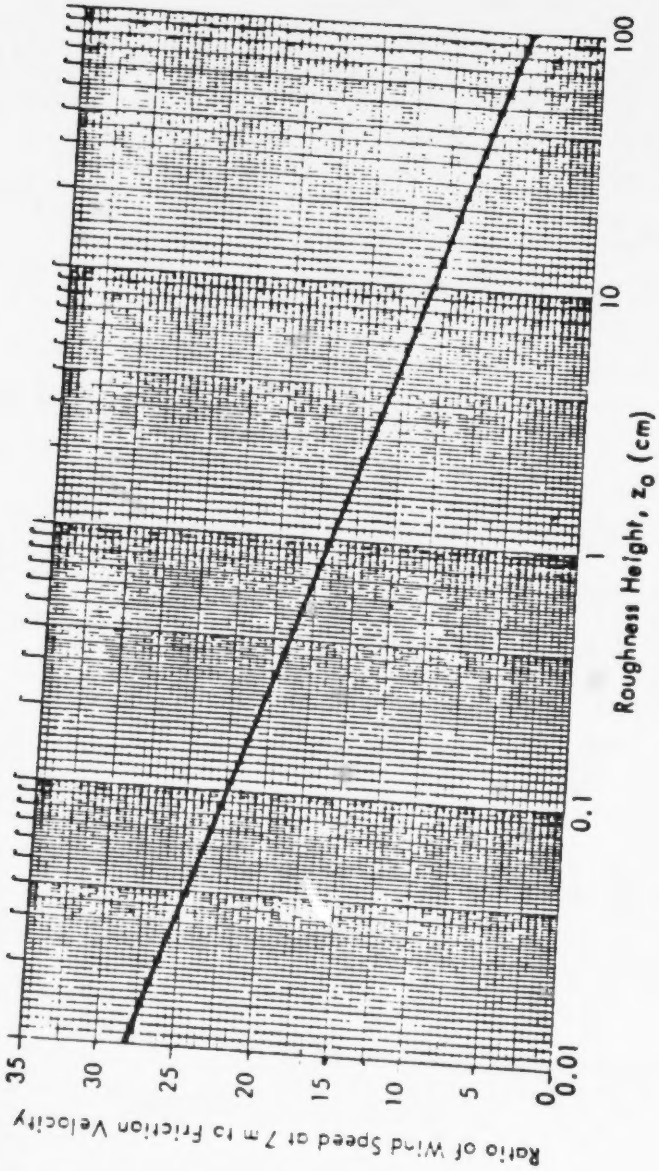


Figure 8-2. Ratio of wind speed at 7m to friction velocity as a function of roughness height

Source: U.S. EPA, 1985a

where:

V = 0
$[u]$ = 3.9 m/s (U.S. EPA, 1985a)
u_t = 7.5 m/s
x = 1.7
$F(x)$ = 0.62

$$E_{10} = 0.036 \cdot (1-0) \cdot (3.9\ m/s/7.5\ m/s)^3 \cdot (0.62)$$
$$= 0.0031\ g/hr/m^2$$

For the second field example, the field is also assumed to be bare, so V is 0. The mean annual wind speed [u] of Nebraska is 4.6 m/s (U.S. EPA, 1985a). The wind erosion friction velocity is assumed to be 50 cm/s and the roughness height is assumed to be 2.0 cm. Rw for a roughness height of 2.0 is 15 as derived from Figure 8-2. Thus, the threshold wind speed for the field in Nebraska is 7.5 m/s.

$$u_t = R_w \cdot F_v$$

where:

R_w = 15
F_v = 50 cm/s

$$u_t = 15 \cdot 50\ cm/s$$
$$= 7.5\ m/s$$

Therefore x is calculated to be 1.4.

$$x = 0.886 \cdot (u_t/[u])$$

where:

u_t = 7.5 m/s
$[u]$ = 4.6 m/s

$$x = 0.886 \cdot (7.5\ m/s/4.6\ m/s)$$
$$= 1.4$$

Thus, $F(x)$ is 1.0 as estimated from the graph in Figure 8-1. E_{10} for the field in Nebraska is calculated by:

$$E_{10} = 0.036 \cdot (1 - V) \cdot ([u]/u_t)^3 \cdot F(x)$$

where:

$V = 0$
$[u] = 4.6$ m/s
$u_t = 7.5$ m/s
$x = 1.4$
$F(x) = 1.0$

$$E_{10} = 0.036 \cdot (1 - 0) \cdot (4.6 \text{ m/s}/7.5 \text{ m/s})^3 \cdot (1.0)$$
$$= 0.0083 \text{ g/hr/m}^2$$

Since the area of the field, the pollutant concentration in the soil and the PM_{10} emission factor are known for each example, the annual emission rates can be calculated using equation 8-1.

For the field in western Florida, the emission rate is calculated by:

$$R_{10} = A \cdot \alpha \cdot E_{10}$$

where:

$A = 10{,}000 \text{ m}^2$
$\alpha = 4.06 \times 10^{-4}$ mg Cd/g soil
$E_{10} = 0.0031 \text{ g/hr/m}^2$

$$R_{10} = 10{,}000 \text{ m}^2 \cdot 4.06 \times 10^{-4} \text{ mg/g} \cdot 0.0031 \text{ g/hr/m}^2 \cdot 1 \text{ hr}/3600 \text{ s}$$
$$= 3.50 \times 10^{-6} \text{ mg/s}$$

The annual emission rate of the contaminant is 3.50×10^{-6} mg/s.

For the Nebraska example, the annual average emission rate of cadmium is calculated by:

$$R_{10} = A \cdot \alpha \cdot E_{10}$$

where:

A = 400,000 m^2
α = 1.39×10^{-4} mg Cd/g soil
E_{10} = 0.0083 $g/hr/m^2$

$$R_{10} = 400{,}000\ m^2 \cdot 1.39\times10^{-4}\ mg/g \cdot 0.0083\ g/hr/m^2 \cdot 1\ hr/3600\ s$$

$$= 1.28\times10^{-4}\ mg/s$$

8.4.3. Worst-Case 24-hr Emission Rate. The worst-case 24-hr emission rate of contaminant on PM_{10} particles (R_{10}) is the product of the area of the field, the pollutant soil concentration, and the emission factor of PM_{10} particles.

$$R_{10} = A \cdot \alpha \cdot E_{10} \qquad \text{(Equation 8-5)}$$

where:

R_{10} = worst-case 24-hr emission rate (mg/s)
A = area of the source (m^2)
α = soil concentration of the contaminant (mg/g)
E_{10} = worst-case 24-hr emission factor ($g/hr/m^2$)

The difference between the worst-case and mean annual emission rate is the emission factor (E_{10}). The worst-case emission factor is derived using the maximum 6-hr wind speed whereas the mean annual emission factor is based on the average wind speed.

For the first example, the field in western Florida is assumed to have an area of 10,000 m^2. The soil concentration of cadmium is assumed to be 4.06×10^{-4} mg/g as previously calculated. For the second example, the field in Nebraska, the area is assumed to be 400,000 m^2. The soil concentration is assumed to equal 1.39×10^{-4} mg/g.

The 24-hr emission factor is calculated by:

$$E_{10} = 0.036 \cdot (1 - V) \cdot [u_{6\text{-}hr}]^3 \qquad \text{(Equation 8-6)}$$

where:

V = fraction of contaminated surface vegetation cover

$[u_{6\text{-}hr}]$ = expected maximum 6-hr mean wind speed during the year (m/s)

The worst-case 24-hr emission factor is based on the expected maximum 6-hr wind speed during the year $[u_{6\text{-}hr}]$. The expected maximum 6-hr mean wind speed during the year is calculated by:

$$[u_{6\text{-}hr}] = [u^+] - 2 \text{ m/s} \qquad \text{(Equation 8-7)}$$

where:

$[u^+]$ = annual fastest mile (m/s)

The annual fastest mile reflects the magnitude of wind gusts; it is speed of the fastest whole mile of wind movement that passed by the 1-mile contact anemometer (U.S. EPA, 1985a).

For the first example, the field is assumed to be bare, so V equals 0. The fastest mile in Tampa, Florida is 22.2 m/s (U.S. EPA, 1985a), so the expected maximum 6-hr mean wind speed is calculated using equation 8-7.

$$[u_{6\text{-}hr}] = [u^+] - 2 \text{ m/s}$$

where:

$[u^+]$ = 22.2 m/s

$$[u_{6\text{-}hr}] = 22.2 \text{ m/s} - 2 \text{ m/s}$$

$$= 20.2 \text{ m/s}$$

Thus, the 24-hr emission factor for the field in western Florida is calculated using equation 8-6.

$$E_{10} = 0.036 \cdot (1 - V) \cdot [u_{6\text{-}hr}]^3$$

where:

$V = 0$
$[u_{6\text{-hr}}] = 20.2$ m/s

$$E_{10} = 0.036 \cdot (1 - 0) \cdot [20.2 \text{ m/s}]^3$$
$$= 297 \text{ g/hr/m}^2$$

For the second example, the field was assumed to be bare, so V equals 0. The fastest mile for North Platte, Nebraska is 27.7 m/s (U.S. EPA, 1985a) so the expected maximum 6-hr mean wind speed during the year is calculated by:

$$[u_{6\text{-hr}}] = [u^+] - 2 \text{ m/s}$$

where:

$[u^+] = 27.7$ m/s

$$[u_{6\text{-hr}}] = 27.7 \text{ m/s} - 2 \text{ m/s}$$
$$= 25.7 \text{ m/s}$$

Thus, the 24-hr emission factor for the field in Nebraska is calculated by:

$$E_{10} = 0.036 \cdot (1 - V) \cdot (u_{6\text{-hr}})^3$$

where:

$V = 0$
$[u_{6\text{-hr}}] = 25.7$ m/s

$$E_{10} = 0.036 \cdot (1 - 0) \cdot (25.7 \text{ m/s})^3$$
$$= 611 \text{ g/hr/m}^2$$

When the area, soil concentration, and E_{10} are known for a particular source, the 24-hr emission rate may be estimated using Equation 8-5. For the field in western Florida, the 24-hr emission rate is calculated by:

$$R_{10} = A \cdot \alpha \cdot E_{10}$$

where:

$A = 10{,}000\ m^2$
$\alpha = 4.06 \times 10^{-4}$ mg Cd/g soil
$E_{10} = 297\ g/hr/m^2$

$$R_{10} = 10{,}000\ m^2 \cdot 4.06 \times 10^{-4}\ mg/g \cdot 297\ g/hr/m^2 \cdot 1\ hr/3600\ s$$
$$= 3.35 \times 10^{-1}\ mg/s$$

For the field in Nebraska, the 24-hr emission rate is calculated by:

$$R_{10} = A \cdot \alpha \cdot E_{10}$$

where:

$A = 400{,}000\ m^2$
$\alpha = 1.39 \times 10^{-4}$ mg Cd/g soil
$E_{10} = 611\ g/hr/m^2$

$$R_{10} = 400{,}000\ m^2 \cdot 1.39 \times 10^{-4}\ mg/g \cdot 611\ g/hr/m^2 \cdot 1\ hr/3600\ s$$
$$= 9.44\ mg/s$$

8.4.4. Results. The results of these calculations indicate that the emission rate of cadmium in resuspended dust may be similar to the stack emission rate of the incinerator ($1.9 \times 10^4\ \mu g/s$) depending on the scenario. For the long-term exposure assessment, the mean annual emission rate for each example is at least 3 orders of magnitude less than the stack emission rate even though "worst-case" situations are assumed. Each field is assumed to have no vegetative cover to provide a conservative estimate of dust release. If vegetation is present, the resulting emission factor and rate will decrease. The mean annual emission rates based on this conservative assumption are 3.50×10^{-6} mg/s (western Florida) and 1.28×10^{-4} mg/s (Nebraska).

For the short-term assessment, the 24-hr emission rate for each example is greater than the stack emission rate. By assuming no vegetative cover is present at either site, the worst case 24-hr emission rates are found to be 0.335 mg/s (western Florida) and 9.44 mg/s (Nebraska).

To determine the extent of human exposure to the resuspended dust, the emission rates, either mean annual emission rate or 24-hour emission rate, must be incorporated into an air dispersion model. The model uses this input along with site-specific meteorology data (i.e., wind speed and cloud cover) to predict ambient air concentrations.

For cases in which the emission rates are less than the stack emission rate and the human health risk posed by inhalation of the pollutants emitted from the stack is small, the analyst may chose not to run the air dispersion model since the ambient air concentrations would be less than or equal to the ambient air concentrations emitted from the combustor. The cancer risk caused by the inhalation of the resuspended pollutants by wind erosion would be less than that caused by the inhalation of the stack emissions.

8.5. CONCLUSION

When determining exposure to pollutants resuspended in the soil, the amount of vegetation, soil concentration, size of the source, wind velocity, moisture content and soil roughness height should be examined. Various methodologies have been developed to determine the soil loss resulting from wind erosion. However in applying these methods to deposited combustor emissions, the analyst should review the approaches discussed in U.S. EPA (1985a), Farino et al. (1983), Sehmel (1980), and Smith et al. (1982) to select the most appropriate methodology for the exposure scenario. The different types of exposure scenarios, whether long-term or short-term, may yield emission rates differing by orders of magnitude.

9. CALCULATING WATER CONCENTRATIONS

9.1. INTRODUCTION

Contaminants associated with particles emitted by combustors are deposited on land and water downwind from the combustor at rates determined by meteorology, terrain and particle physics. Following deposition, these contaminants may be dissolved, repartitioned or degraded. They may be transported over the ground in runoff to a surface water body or infiltrate the ground and recharge the groundwater. This chapter focuses on the contaminant concentration found in surface water bodies, collected precipitation and groundwater.

9.2. SURFACE WATER

The aim of the Surface Water Model is to estimate the concentration of a given contaminant in a water body following deposition onto a watershed. This model was developed for non-volatile compounds. Estimated contaminant levels will be the result of runoff and soil erosion, as well as direct deposition of particulate matter onto the water body. The model is designed to accommodate both long-term and short-term loading scenarios. The surface water model follows a three-tiered approach beginning with a simple, extremely conservative calculation (Tier 1), and proceeding to more detailed calculations when necessary (Tier 2 and Tier 3). Tier 2 and 3 calculations differ only in the use of literature input values (Tier 2) vs. site-specific (Tier 3) input variables.

The methodology derived to calculate risks from the surface runoff pathway originally was developed to evaluate impacts from the land application of municipal wastewater sludge. A detailed discussion of the model is available in the document entitled Development of Risk Assessment Methodology for Land Application and Distribution and Marketing of Municipal Sludge (U.S. EPA, 1989b).

9.2.1. Overview of the Model. The Surface Runoff Model is used to calculate the concentration of contaminant, both dissolved and adsorbed to suspended particles, that accumulates in a surface water body both from deposition directly onto the water body and deposition onto the watershed followed by surface runoff. These predicted concentrations are used to estimate human risk that may result from drinking water, swimming and bathing in, or eating fish from a contaminated surface water body.

Tier 1 of the Surface Runoff Model is a conservative screening method that assumes that all contaminant deposited onto a watershed/water body is transported to the receiving water body during the loading period (i.e., 1 year for long-term loading and 24 hours for short-term loading). The contaminant concentration in the receiving water is calculated from the annual mass of contaminant in the fallout per unit area, the total area of the watershed, and dilution volume of the water body under consideration (see discussion in Section 9.2.2). The Tier 1 water concentration is used to calculate daily intake due to water ingestion, dermal absorption, and fish ingestion. These daily intakes are either compared with a reference dose for chronic oral exposure (RfD_o) or used to calculate a risk level as described in Chapter 15. If these analyses show unacceptable excess risk, a Tier 2 or Tier 3 analysis is required to determine a more reliable estimate of the receiving water concentration.

Tier 2 and 3 calculations provide more detailed analysis of the fate of contaminant in the watershed. Tier 2 and 3 calculations are identical; they differ only in origin of input values. Tier 2 relies on input values found in the literature. Tier 3 is based on site-specific data for input parameters. In Tier 2 or 3, contaminant concentration in water is calculated for long-term loading (i.e., 1 year) or short-term loading (i.e., 24 hours).

For long-term loading, the model assumes all contaminants entering the receiving water are absorbed to eroded particles and do not partition between soil particles and water. The contaminant concentration in water is calculated from the mass of contaminant per area of soil, the distribution coefficient (K_d), the area of soil eroded to the water body, and the appropriate dilution volume of the water body in question.

For short-term loading, the model assumes that the contaminant enters the receiving water body by direct deposition, absorbed to eroded soil particles, and dissolved in the runoff water. Once the contaminants enter the receiving water, they repartition between the absorbed and dissolved state. The contaminant concentration in water, dissolved and adsorbed, is calculated from the total mass of chemical in the water body, the fraction of chemical dissolved or adsorbed, and the appropriate dilution volume of the water body in question.

A number of assumptions about runoff generation and subsequent mixing in the receiving water were required to formulate the risk-based methodology. The key assumptions are listed in Table 9-1, with a discussion of their impact on the methodology. Because the science is not exact, the assumptions generally are conservative (i.e., overpredict contaminant concentrations). In some instances, however, the nature of the effect of a given assumption may vary with site-specific considerations.

9.2.2. Calculating Contaminant Concentrations for Tier 1. In Tier 1, all contaminant deposited onto a watershed is assumed to be transported to the receiving water in the loading period. The total mass flux of contaminant is distributed among the watersheds in which the fallout is deposited. The appropriate dilution volume then is used to calculate the resulting concentration in the receiving water:

$$Ci_x = [Fa \cdot (WA_L + WA_W)]/Vf_x \cdot 10^6\ \mathrm{mg/kg} \cdot \mathrm{m}^3/10^3\ \mathrm{L}$$

(Equation 9-1)

TABLE 9-1

Surface Water Methodology Assumptions

Functional Area	Assumption	Ramifications
Long-Term Concentrations:		
Tier 1	All contaminant deposited on an annual basis is transported to the receiving water in a dissolved form.	Provides an extremely conservative estimate since no losses are considered.
Tier 2 and 3 General	Loadings to the receiving water can be described as a function of solids loading.	Mechanistically inappropriate for contaminants with low partition coefficients.
Source Area	Facility operates over a period sufficient for surface soil levels to reach equilibrium where annual losses equal annual inputs.	Overpredicts contaminant loading by ignoring loss mechanisms other than those used in formulation, namely, runoff and infiltration.
	No settling of particles in the deposition zone, gross erosion reaches the edge of field.	Maximizes contaminant loss.
Short-Term Concentrations:		
Tier 1	All contaminant emitted in a year is lost in a single runoff event.	Provides extremely conservative estimate with no provision for losses or incomplete mobilization.
Tier 2 and 3 Source Area	No kinetically limited release of contaminant from residual/soil mixture; i.e., total contaminant concentration is fully equilibrated into adsorbed and dissolved phases.	Maximizes contaminant concentration available for runoff in dissolved phase, thereby maximizing loadings.

TABLE 9-1 (cont.)

Functional Area	Assumption	Ramifications
Stream	Stream flow is unchanged by the storm unless arterial velocity data are available from the hydrograph.	Overestimates stream concentration, since the storm will increase stream flow.

where:

x = long- or short-term scenario
Ci_{Sh1} = concentration of contaminant in the receiving water for the short-term loading scenario (mg/L)
Ci_{Lg1} = concentration of contaminant in the receiving water for the long-term loading scenario (mg/L)
Fa = annual mass of contaminant in fallout per unit area (kg/km^2-yr)
WA_L = land area of watershed receiving fallout (km^2)
WA_W = area of water receiving fallout (km^2)
Vf_x = long-term dilution volume for water body under consideration (m^3/yr)
  Vf_x(river) = mean annual flow (m^3/yr)
  Vf_x(lake) = mean lake volume (m^3/yr) + mean annual outflow from lake (m^3/yr)
  Vf_x(estuary) = mean annual estuary flow (m^3/yr)

Equation 9-1 is appropriate for the long-term loading scenario. For a short-term loading scenario, Vf_x is replaced with VS_x, the short-term dilution volume. VS_x for the river and estuary can be set to Vf_x times the short-term time frame (in years); however ideally, VS_x for a river or estuary would be obtained from the hydrograph for the storm, if it is known. VS_x for the lake equals the mean lake volume.

This very conservative calculation accounts for no losses during transport and, therefore, overpredicts contaminant levels. If these overpredictions do not exceed health-based criteria (such as the reference dose, RfD), the risks are deemed acceptable and no further analysis is required. If the predictions exceed criteria, a more detailed Tier 2 or 3 analysis is required to determine probable receiving water concentrations. The predicted water concentrations are also used for determination of excess risk (ER) for carcinogens. The risk characterization for this exposure is discussed in Chapter 15.

9.2.3. Calculating Contaminant Concentrations for Tier 2 or 3

9.2.3.1. LONG-TERM EXPOSURE -- For the long-term loading scenario, receiving water contaminant concentration is calculated from estimates of the contaminant mass per unit area of watershed, the rate of sediment transport to the receiving water, and the appropriate dilution for the water body in question (defined in Equation 9-1):

$$Ci_{Lg23} = [X_e \cdot WA_L \cdot Mm/(Vf_x \cdot BD \cdot Z)] \cdot 10^5\ \mathrm{cm/km} \cdot \mathrm{km}^3/10^9\ \mathrm{m}^3 + [Mw \cdot 10^6\ \mathrm{mg/kg} \cdot \mathrm{m}^3/10^3\ \mathrm{L}] \qquad \text{(Equation 9-2)}$$

where:

Ci_{Lg23} = concentration of contaminant in the receiving water for the long-term loading scenario (mg/L)

X_e = sediment loss rate per unit area watershed over time (kg/km^2-yr, see Equation 9-3)

WA_L = land area of watershed receiving fallout (km^2)

Mm = maximum contaminant mass per area of soil (kg/km^2, see Equation 9-4)

Vf_x = long-term dilution volume for water body under consideration (m^3/yr)

BD = bulk density of soil (kg/m^3)

Z = depth of mixing (cm)

Mw = maximum contaminant mass per volume of water (kg/km^3, see Equation 9-5)

The rate of sediment loss to the receiving water is computed using the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978):

$$X_e = R \cdot K \cdot LS \cdot C \cdot P_s \cdot 907.18\ \mathrm{kg/ton} \cdot 1/4.07\times10^{-3}\ \mathrm{acre/km}^2 \qquad \text{(Equation 9-3)}$$

where:

X_e = sediment loss rate per unit area watershed over time (kg/km^2-yr)
R = "erosivity" factor (yr^{-1})
K = "erodability" factor (tons/acre)
LS = "topographic or slope length" factor (unitless)
C = "cover management" factor (unitless)
P_s = "supporting practice" factor (unitless)

Guidelines for selection of input factor values for the USLE in each watershed are provided in Wischmeier and Smith (1978) and in the runoff discussion of the U.S. EPA (1989b). The input parameters used are site-specific values. X_e is an average annual value and is obtained by multiplying estimates of soil loss from a rainfall and runoff for a geographic area by a series of ratios. R is the erosion potential for average annual rainfall at a given location. K is an experimentally determined value using the predominant soil type as a guide in obtaining the value for K. LS takes into account the effect of slope length and steepness, while C looks at the vegetative cover, crop sequence, crop rotation and tilling practices. P_s looks at the agricultural techniques such as contouring and terracing.

In order to estimate contaminant mass per area of soil (Mm), the combustors are assumed to be in operation long enough for a steady state concentration to be reached. By definition, these conditions will prevail when soil levels are high enough for the sum of zeroth and first-order losses to equal the annual additional contaminants in fallout. Maximum soil contaminant mass per area of soil is calculated as:

$$Mm = Fa \cdot [1 - \exp(k_1 \cdot timef)/k_1] \quad \text{(Equation 9-4)}$$

where:

Mm = maximum contaminant mass per area of soil (kg/km^2)

Fa = annual fallout rate for contaminant losses (kg/km^2-yr)

k_1 = first-order loss rate (see Equation 9-6) (yr^{-1})

timef = time span in which analysis will be carried out, typically the life of the combustor and its replacements (yr)

Since contamination from the combustion facility enters the water body from both runoff and direct fallout, the contribution from direct deposition must be accounted for in the model. An equation similar to Equation 9-4 is used to estimate the accumulation of the contaminant in the water body from direct deposition:

$$Mw = Fa \cdot [WA_w/Vf_x] \quad \text{(Equation 9-5)}$$

where:

Mw = maximum contaminant mass per volume of water (kg/m^3)

Fa = annual fallout rate for contaminant losses (kg/km^2-yr)

WA_w = area of water receiving fallout (km^2)

Vf_x = long-term dilution volume for water body under consideration (m^3/yr)

The first-order loss rate, k_1, can be calculated by adding the loss rates due to infiltration (k_{1I}), erosion (k_{1E}), and degradation (k_{1D}); the equations for each of these loss rates is presented below:

$$k_1 = k_{1I} + k_{1E} + k_{1D} \quad \text{(Equation 9-6)}$$

$$k_{1I} = \frac{I_R}{\theta \cdot Z \cdot [1 + (BD \cdot K_d / \theta)]} \qquad \text{(Equation 9-7)}$$

where:

k_{1I} = First-order loss rate for infiltration (yr^{-1})
I_R = Infiltration rate (cm/yr); set default value equal to 1 cm/yr
θ = Volumetric water content of the soil (mL/cm^3)
Z = Depth of incorporation (cm)
BD = Soil bulk density (kg/m^3)
K_d = Distribution coefficient (m^3/kg)

$$k_{1E} = \frac{X_e}{BD \cdot Z} \cdot \left[\frac{K_d \cdot BD}{\theta + (K_d \cdot BD)}\right] \cdot 10^5 \text{ cm/km} \cdot \text{km}^3/10^9 \text{ m}$$

(Equation 9-8)

where:

k_{1E} = First-order loss rate for erosion (yr^{-1})
X_e = sediment loss rate per unit area watershed over time (kg/km^2-yr)
Z = Depth of incorporation (cm)

$$k_{1D} = \ln 2/t_{1/2} \qquad \text{(Equation 9-9)}$$

where:

k_{1D} = first-order loss rate for degradation (yr^{-1})
$t_{1/2}$ = contaminant half-life due to degradation in soil (yr)

In general, volatilization would also represent a first-order loss mechanism. For MWC particulate fallout, however, it is assumed that volatile species will not be present in the solids settling from the atmosphere. Also, when field measurements are available for deriving degradation rates, they may reflect all first-order losses (k_1) and not just degradation (k_{1D}). Therefore, care must be taken in selecting input values.

9.2.3.2. SHORT-TERM EXPOSURE -- For short-term loading scenarios, the model calculates the dissolved and adsorbed contaminant concentration in receiving water after a single storm event.

To calculate the concentration of chemical present in either dissolved or adsorbed form in the water body, the mass of chemical per unit volume of water is divided by the appropriate dilution volume of the water body in question:

$$CHEM_w = CHEM_d/VS_x \cdot m^3/10^3\ L \qquad \text{(Equation 9-10)}$$

and

$$CHEM_{s1} = CHEM_a/X_s \qquad \text{(Equation 9-11)}$$

$$CHEM_{s2} = CHEM_a/VS_x \cdot m^3/10^3\ L \qquad \text{(Equation 9-12)}$$

where:

$CHEM_w$ = mass of dissolved chemical per unit volume of water (mg/L)

$CHEM_d$ = mass of chemical present in the water body in the dissolved form (mg)

VS_x = appropriate dilution volume for the watershed under consideration over the duration of the storm event (m^3)

$CHEM_{s1}$ = mass of adsorbed chemical per mass of sediment in the water body (mg/kg)

$CHEM_a$ = mass of chemical present in the water body in the adsorbed form (mg)

X_s = mass of sediment in the water body (washed off from the watershed) (kg)

$CHEM_{s2}$ = mass of adsorbed chemical per unit volume of water (mg/L)

Vf_x = appropriate dilution volume for the watershed under consideration - use Vf_x when calculating $CHEM_w$ for long-term exposure (m^3)

To determine the mass of chemical in the water body in either the dissolved or adsorbed form, the following equations must be solved:

$$CHEM_d = TOTCHEM \cdot \alpha_d \cdot 10^6 \text{ mg/kg} \quad \text{(Equation 9-13)}$$

and

$$CHEM_a = TOTCHEM \cdot \alpha_a \cdot 10^6 \text{ mg/kg} \quad \text{(Equation 9-14)}$$

where:

$CHEM_d$ = mass of chemical present in the water body in the dissolved form (mg)
TOTCHEM = total mass of the chemical in the water body (kg, see Equation 9-19)
αd = fraction of chemical dissolved (unitless, see Equation 9-15)
$CHEM_a$ = mass of chemical present in the water body in the adsorbed form (mg)
α_a = fraction of chemical adsorbed (unitless, see Equation 9-16)

In the remainder of this section, the calculations needed to determine the fractions of dissolved and adsorbed chemical will be presented first followed by the calculations needed to determine the total mass of the chemical.

The fraction of chemical present in either the dissolved or adsorbed state can be calculated by the following equations:

$$\alpha_d = \frac{1}{1 + (K_d \cdot C_s)} \quad \text{(Equation 9-15)}$$

and

$$\alpha_a = \frac{K_d \cdot C_s}{1 + (K_d \cdot C_s)} \quad \text{(Equation 9-16)}$$

where:

α_d = fraction of chemical dissolved (unitless)
K_d = distribution coefficient for the contaminant in the soil water system (m^3/kg)
C_s = mean sediment concentration in the water body (kg/m^3, see Equation 9-17)
α_a = fraction of chemical adsorbed (unitless)

The concentration of sediment in the water body during a storm event can then be calculated using appropriate dilution volume of the water body because the volume of soil that washes off the watershed is negligible compared with the dilution volume. The mean sediment concentration is calculated by the following equation:

$$C_s = X_s/VS_x \qquad \text{(Equation 9-17)}$$

where:

C_s = mean sediment concentration in the water body (kg/m^3)
X_s = mass of sediment in the water body (washed off from the watershed) (kg, see Equation 9-27)
VS_x = the appropriate dilution volume for water body under consideration over the duration of the storm event (m^3)

The total mass of the chemical in the water body is equal to the amount of chemical present in the dissolved form in the watershed plus the mass of chemical in the adsorbed form in the watershed, or:

$$TOTCHEM = P_{xt} + P_{qt} + P_{ft} \qquad \text{(Equation 9-19)}$$

where:

TOTCHEM = total mass of the chemical in the water body (kg)
P_{xt} = mass of chemical entering water body in adsorbed form (kg, see Equation 9-21)
P_{qt} = mass of chemical entering water body in dissolved form (kg, see Equation 9-22)
P_{ft} = mass of chemical deposited directly onto the water body (kg, see Equation 9-20)

To determine the amount of chemical that is deposited directly onto the watershed the following equation must be solved:

$$P_{ft} = Fa \cdot T_f \cdot WA_w \qquad \text{(Equation 9-20)}$$

where:

P_{ft} = mass of chemical deposited directly onto the water body (kg)
Fa = annual mass of contaminant in fallout per unit area (kg/km^2-yr)
T_f = time frame of the model, i.e., 0.002738 years for the short-term loading scenario and 1 year for the long-term loading scenario (yr)
WA_w = area of water receiving fallout (km^2)

The mass of contaminant entering the water body in adsorbed or dissolved form can be calculated using the following equations:

$$P_{xt} = [X_s/(BD \cdot Z)] \cdot Aa \cdot 10^5 \text{ cm/km} \cdot 10^9 \text{ m}^3/\text{km}^2 \qquad \text{(Equation 9-21)}$$

and

$$P_{qt} = [D_R/(R_t + M_t)] \cdot Da \cdot WA_L \qquad \text{(Equation 9-22)}$$

where:

P_{xt} = mass of contaminant entering water in adsorbed form (kg)
X_s = mass of sediment in the water body (washed off from the watershed) (kg)
BD = bulk density of the soil (kg/m^3)
Z = depth to which tilling is practiced (cm), the default value should be equal to 1
Aa = adsorbed contaminant mass per unit of land in watershed (kg/km^2, see Equation 9-23)
P_{qt} = mass of contaminant entering water in dissolved form (kg)
D_R = depth of runoff in the watershed (cm, see Equation 9-25)
R_t = depth of total rainfall for the storm event (cm)
M_t = depth of snow melt for the storm event (cm), set default value equal to zero
Da = dissolved contaminant mass per unit of land in watershed (kg/km^2, see Equation 9-24)
WA_L = land area of watershed receiving fallout (km^2)

If the area under consideration is an agricultural area that is likely to be tilled, soil depth is assumed to be 20 cm. In all other cases, soil depth is assumed to be 1 cm.

To determine the mass of dissolved and adsorbed contaminant entering the water body, the total mass of contaminant, Mm, deposited on land surface must be partitioned between the adsorbed portion, Aa, and the dissolved portion, Da. These are derived as:

$$Aa = [1/(1+(\theta/K_d \cdot BD))] \cdot Mm \quad \text{(Equation 9-23)}$$

and

$$Da = [1/(1+(K_d \cdot BD/\theta))] \cdot Mm \quad \text{(Equation 9-24)}$$

where:

Aa = adsorbed contaminant mass per unit of land in watershed (kg/km^2)
θ = volumetric water content of the top cm of soil (mL/cm^3)
K_d = distribution coefficient for contaminant in soil water system (m^3/kg)
BD = bulk density of the soil (kg/m^3)
Mm = maximum contaminant mass per unit area of soil (kg/km^2)
Da = dissolved contaminant mass per unit area of land in watershed (kg/km^2)

The runoff depth of carrying dissolved contaminant to the water body is estimated by the following equation:

$$D_R = \frac{(R_t + M_t - 0.2S)^2}{(R_t + M_t + 0.8S)} \qquad \text{(Equation 9-25)}$$

where:

D_R = depth of runoff in the watershed (cm)
R_t = depth of total rainfall for the storm event (cm)
M_t = depth of snow melt during the storm event (cm)
S = the watershed retention parameter (cm)

The watershed retention factor (S) is calculated from the Soil Conservation Service (SCS) runoff curve number (CN) according to the following equation:

$$S = 2.54 \cdot [(1000/CN)-10] \qquad \text{(Equation 9-26)}$$

The mass of sediment eroded from the watershed to receiving water during a storm event is estimated with the Modified Universal Soil Loss Equation (MUSLE):

$$X_s = 2.04\text{x}10^6 \cdot (Q \cdot q_p)^{0.56} \cdot K \cdot LS \cdot C \cdot P$$

where: (Equation 9-27)

X_s = mass of sediment in the water body (washed off from the watershed) (kg)
Q = volume of runoff (km^2-cm, see Equation 9-28)
q_p = peak runoff (m^3/s, see Equation 9-29)
K = "erodability" factor (tons/acre)
LS = "topographic or slope length" factor (unitless)
C = "cover management" factor (unitless)
P = "supporting practice" factor (unitless)

Again, selection of K, LS, C and P is discussed elsewhere (Wischmeier and Smith, 1978; U.S. EPA, 1989b). This equation is an empirical relationship and the units must be consistent with those shown above.

The storm runoff volume is calculated from the runoff depth according to:

$$Q = WA_L \cdot D_R \qquad \text{(Equation 9-28)}$$

where:

Q = volume of runoff (km^2-cm)
WA_L = land area of the watershed receiving fallout (km^2)
D_R = depth of runoff from the storm event (cm)

A trapezoidal hydrograph is assumed so that the peak runoff rate can be calculated as:

$$q_p = \frac{WA_L \cdot D_R \cdot R_t}{T_r \cdot (R_t - 0.2S)} \cdot \text{hr}/3.6\text{x}10^3\ \text{s} \cdot \text{km}/10^5\ \text{cm} \cdot 10^9\ \text{m}^3/\text{km}^3$$

(Equation 9-29)

where:

q_p = peak runoff rate (m^3/s)
WA_L = land area of the watershed (km^2)
D_R = depth of runoff in the watershed (cm)
R_t = depth of total rainfall for the storm event (cm)
T_r = duration of the storm event (hrs)
S = water retention parameter (cm)

Example calculations for this model are presented in Section 9.5.

9.3. PRECIPITATION

In some remote areas, rain and/or snowfall are collected off surfaces and stored in cisterns to be used as a source of drinking water. Individuals in these areas may be exposed to pollutants deposited on surfaces downwind from the combustor from which the water is collected.

9.3.1. Water Concentration (Wc). The water concentration of pollutants (mg/L) found in collected precipitation can be determined as follows:

$$Wc = \frac{Dy}{P} \qquad \text{(Equation 9-30)}$$

where:

Wc = water concentration (mg/L)
Dy = total deposition ($g/m^2/yr$)
P = annual precipitation (mm/yr)

The value for water concentration obtained from this equation is used as an input variable in calculations to determine the human health risk associated with dermal absorption and ingestion of water.

9.3.1.1. Total Deposition (Dy). The total deposition is from both wet and dry site-specific deposition rates, which are determined by air-dispersion and deposition modeling. Methods for determining areal-averaged deposition rates are described in Chapter 3.

9.3.1.2. Annual Precipitation (P). Site-specific values for the annual precipitation are available in Baes et al. (1984).

9.4. GROUNDWATER

Once contaminated particles emitted from combustors are deposited on surfaces downwind from the facility, the contaminants may become dissolved in precipitation such as mist, rain, hail, sleet and snow. The dissolved portion can either form surface runoff as described in Section 9.2 or infiltrate the ground as leachate and recharge the groundwater. Individuals will be exposed if the contaminated groundwater is used as a source of drinking water.

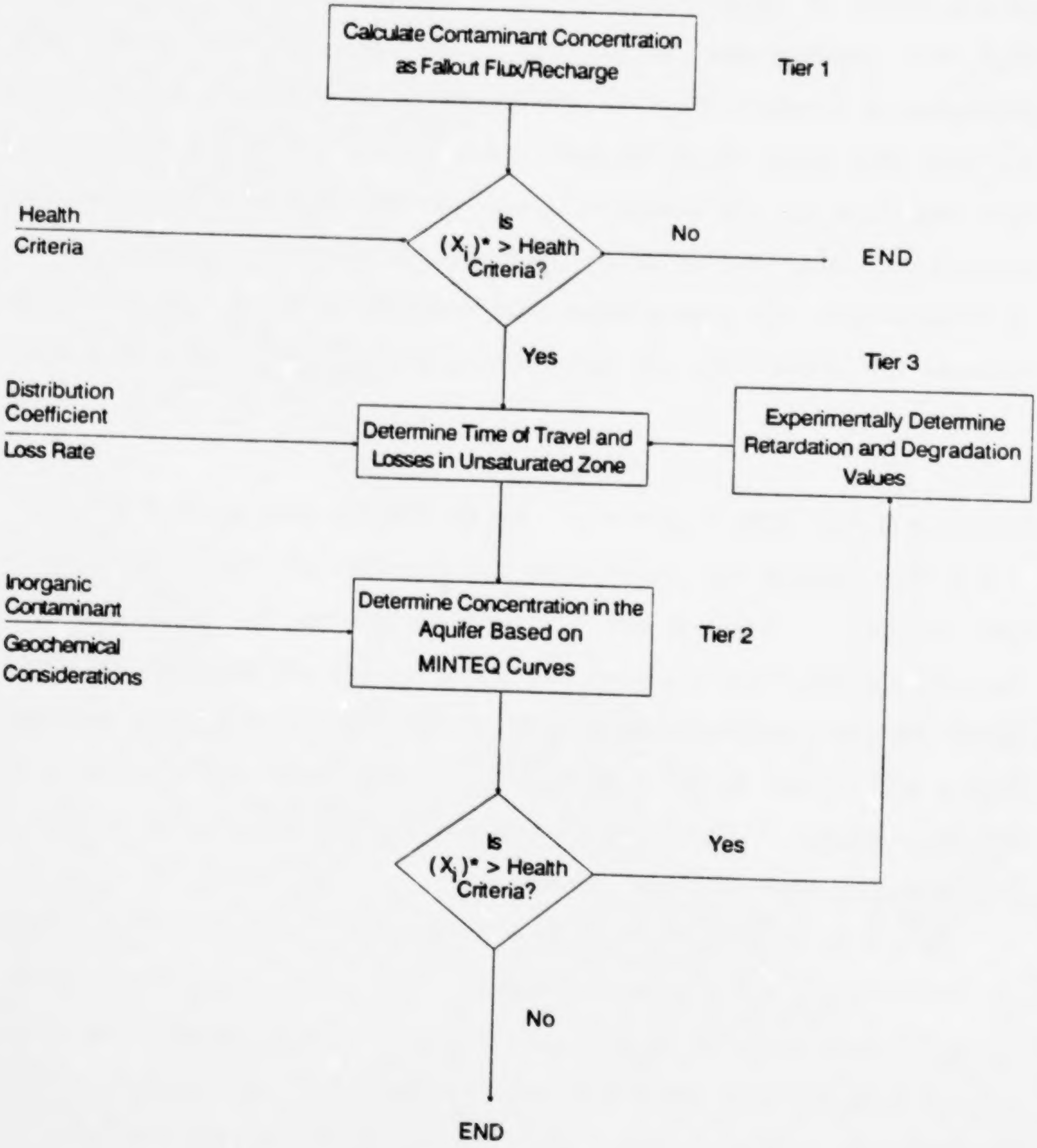
In determining the risk from exposure to groundwater, a number of factors affect leachate generation and subsequent transport of contaminants in the unsaturated zone. Some factors that affect leachate generation are the physicochemical characteristics of the emitted chemical, the magnitude of precipitation and environmental transport. Infiltration of the leachate can be affected by thickness of the saturated layer, moisture content of the soil, degree of compaction, macrostructure of soil, vegetative cover, temperature, and entrapped air (U.S. EPA, 1988a). For metals, the contaminant concentration in the groundwater is affected by geochemical reactions. For organics, the contaminant concentration is altered by degradation as it travels through the unsaturated zone.

A Groundwater Infiltration Model (GIM) was developed to model contaminant movement to groundwater. The model, described in U.S. EPA (1987c), was used to make preliminary determinations of the potential for groundwater contamination under worst-case circumstances by several inorganic and organic contaminants. These findings indicated very limited potential for contamination. Therefore, further evaluation of this exposure route is considered unnecessary. This section summarizes the GIM and the results of the preliminary calculations.

9.4.1. Overview of the Model. The GIM is formulated in three successive tiers (see Figure 9-1), which begin with simple but very conservative estimates and proceed to more detailed analyses if the first tiers predict unacceptable risks. Groundwater contaminant concentrations (absorbed and dissolved fractions) predicted by the model are compared with health-based water criteria to ascertain if a potential human health risk exists.

FIGURE 9-1

Logic Flow for Groundwater Pathway Evaluation


Calculate Contaminant Concentration as Fallout Flux/Recharge
Tier 1
Health Criteria
Is $(X_i)^*$ > Health Criteria?
No
END
Yes
Tier 3
Distribution Coefficient
Loss Rate
Determine Time of Travel and Losses in Unsaturated Zone
Experimentally Determine Retardation and Degradation Values
Inorganic Contaminant
Geochemical Considerations
Determine Concentration in the Aquifer Based on MINTEQ Curves
Tier 2
Is $(X_i)^*$ > Health Criteria?
Yes
No
END

* (X_i) = Concentration of Contaminant i

9.4.1.1. TIER 1 -- Tier 1 is an extremely conservative approach in which projected leachate concentrations are compared with health-based criteria. Leachate concentrations are predicted on the basis of annual fallout and recharge rates. It is assumed that soil contaminant levels will increase over time until the contaminant concentration in the leachate is sufficiently high to deplete the input from fallout each year. If the contaminant concentration in the leachate is below the relevant health-based criterion, the contaminant can be eliminated from further consideration. If the contaminant concentration in the leachate exceeds the criterion, the contaminant is carried forward to a Tier 2 or 3 analysis.

9.4.1.2. TIER 2 and 3 -- As in Tier 1, the Tier 2 and Tier 3 analysis begins by calculating the contaminant concentration in the leachate. Tier 2 or 3, however, allows for site-specific inputs to predict dispersion, degradation, and retardation effects, which reduce resultant exposure levels. The difference between Tier 2 and 3 lies in the number of input values that are determined experimentally. In Tier 3, degradation rates and retardation coefficients are measured directly.

The initial step in Tier 2 or 3 is to define the concentration of contaminant in recharge water. As with Tier 1, this is accomplished by assuming that equilibrium will be reached when the inputs from fallout each year are transported away from the soil by leachate. Subsequently, the time required for the leachate to move downward through the unsaturated zone to the aquifer is determined using factors such as field moisture content, depth to the saturated zone and recharge rate. For multiple layer systems, a travel time to the aquifer is calculated for each layer and the total summed across the unsaturated zone. The total time it takes for the leachate to reach the aquifer and the total depth of the unsaturated zone are used to derive the equivalent velocity of the leachate. The velocity of contaminant traveling through the unsaturated zone can be retarded if the contaminant interacts with the soil in transit.

9.4.2. Determining Leachate Contaminant Concentration. The U.S. EPA has used the GIM to evaluate possible health effects from exposure to groundwater contaminated with several metals and organics emitted by municipal waste combustors (U.S. EPA, 1987c). Exposures were modeled for a facility in Hampton, Virginia and a hypothetical model plant in western Florida, chosen to represent a reasonable worst-case scenario at an existing facility and a scenario for planned facilities, respectively. Tables 9-2 and 9-3 show the deposition rate of emissions for these facilities when either a fabric filter or electrostatic precipitator were used as control technology in the U.S. EPA analysis. The resulting concentration of the leachate in the aquifer and time for the leachate to reach the aquifer for these scenarios are also shown. For all the chemicals, the groundwater pathway was an insignificant source of exposure as further described below.

Table 9-2

Groundwater Infiltration Model for Organics

Model Values	Chlorinated Benzenes	Formaldehyde	Polychlorinated Biphenyls	Polychlorinated Dibenzo-p dioxins
Deposition Rate (Fa) (kg/ha-yr) Hampton, VA				
Fabric Filter	4.48×10^{-4}	4.91×10^{-3}	8.10×10^{-6}	1.07×10^{-5}
Electrostatic Precipitator (ESP)	6.53×10^{-3}	7.02×10^{-2}	1.18×10^{-4}	1.57×10^{-4}
Hypothetical Model Plant in Florida				
Fabric Filter	6.37×10^{-5}	1.77 x 10-2	1.49×10^{-6}	3.33×10^{-7}
Electrostatic Precipitator (ESP)	9.33×10^{-4}	2.67×10^{-1}	2.23×10^{-5}	5.00×10^{-6}
Concentration of Leachate Contaminant (mg/L) (tier 2) Hampton, VA				
Fabric Filter	4.17×10^{-24}	0	0	0
Electrostatic Precipitator (ESP)	6.07×10^{-23}	0	0	0
Hypothetical Model Plant in Florida				
Fabric Filter	3.13×10^{-22}	0	0	0
Electrostatic Precipitator (ESP)	4.58×10^{-21}	0	0	0
Time to Reach Aquifer (yrs) Hampton, VA	4,070	25.1	116,000	598,000
Hypothetical Model Plant in Florida	2,360	14.7	80,100	366,000

Table 9-3

Groundwater Infiltration Model for Metals*

Model Values	Beryllium	Chromium	Lead	Mercury	Nickel
Deposition Rate (Fa) (mg/m^2-yr) Hampton, VA					
Fabric Filter	8.53×10^{-7}	1.31×10^{-2}	2.30×10^{-1}	1.09×10^{-2}	1.09×10^{-2}
Electrostatic Precipitator (ESP)	2.33×10^{-6}	3.76×10^{-2}	6.40×10^{-1}	1.71×10^{-1}	3.10×10^{-2}
Hypothetical Model Plant in Florida					
Fabric Filter	1.34×10^{-4}	1.29×10^{-2}	3.52×10^{-1}	6.83×10^{-1}	3.53×10^{-2}
Electrostatic Precipitator (ESP)	3.78×10^{-4}	3.62×10^{-2}	9.93×10^{-1}	8.98×10^{-1}	9.88×10^{-2}
Concentration of Leachate Contaminant (mg/L) Hampton, VA					
Fabric Filter	2.24×10^{-7}	3.44×10^{-3}	6.04×10^{-2}	1.4×10^{-2}	2.86×10^{-3}
Electrostatic Precipitator (ESP)	6.12×10^{-7}	9.87×10^{-3}	1.68×10^{-1}	1.8×10^{-2}	8.13×10^{-3}
Hypothetical Model Plant in Florida					
Fabric Filter	5.36×10^{-5}	5.16×10^{-3}	1.41×10^{-1}	6.9×10^{-2}	1.41×10^{-2}
Electrostatic Precipitator (ESP)	1.51×10^{-4}	1.45×10^{-2}	3.97×10^{-1}	9.1×10^{-2}	3.95×10^{-2}
Time to Reach Aquifer (yrs) Hampton, VA	27,800	3,386	17,900	14,000	2,100
Hypothetical Model Plant in Florida	73,300	3,300	17,400	14,000	1,560

*Tic. [illegible]

9.4.2.1. ORGANICS -- For organic chemicals (Table 9-2), the concentration in the leachate (Tier 2) reaching the aquifer was negligible. Degradation of organics in the unsaturated zone was the primary reason that only trace amounts of the organic contaminants reached the aquifer. Furthermore, except in a few cases, these contaminants took an extremely long time (thousands of years) to reach the aquifer. In the instances where the organic compound was predicted to reach the aquifer in <1000 years, the concentration was predicted to be so small that it was estimated as zero.

9.4.2.2. METALS -- Metal contaminants (Table 9-3) in the leachate do not degrade; however, they move more slowly than organic contaminants because of adsorption and precipitation reactions with soil (Freeze and Cherry, 1979). Although metal concentration in the aquifer may be above the health-based criteria (as with cadmium), a significant amount of the metal is adsorbed to suspended solids, which can be filtered from water. Therefore, these metals would not contribute to human exposure. Furthermore, the model predicted thousands of years before the estimated concentrations would reach the aquifer.

9.4.3. Conclusion. Contaminants in combustor emissions may dissolve, infiltrate the ground and become available for recharging the groundwater. However, as shown by the U.S. EPA analysis discussed above, these contaminants degrade and move slowly through the unsaturated zone to the aquifer. Thus, groundwater is not likely to be a significant route for human exposure to contaminants from deposited combustor emissions. Risk assessors may want to determine contaminant concentration in groundwater using site-specific values to assess the contribution of groundwater to human exposure at a particular site.

9.5. EXAMPLE CALCULATIONS

This section illustrates the methods for determining benzo(a)pyrene and cadmium concentrations in surface water (Surface Water Model, Section 9.2) for three different exposure scenarios: A, B and C. In determining the water concentration of a contaminant, the deposition area used differed in the case of each exposure scenario. In Scenario A, the area of the watershed from the incinerator is within a 50 km radius. In Scenario B, the watershed is within a 5 km radius and in Scenario C the watershed is at the point of maximum deposition at 0.2 km, as discussed in Chapter 2. The scenarios also differ in the operating lifetime of the incinerator. A summary of the calculations for all three scenarios can be found in Table 9-6 for benzo(a)pyrene, and Table 9-9 for cadmium.

9.5.1. Benzo(a)pyrene. The values chosen for the input variables of the Surface Runoff Model are given in Table 9-4. The concentration of a contaminant in the water is a function of its deposition rate (Fa) and the lifetime of the facility (timef). The

TABLE 9-4

Input Variables for Determination of Benzo(a)Pyrene Water Concentrations

Variable	Value
WA_L (watershed area) (km^2)	15.0
WA_W (water body area) (km^2)	1.0
Vf_x (m^3/yr)	3.15×10^7
BD (bulk soil density) (mg/m^3)	1500
Z (soil till depth) (cm)	1.0
R (yr^{-1})	400.0
K (tons/acre)	0.21
LS (unitless)	0.179
C (unitless)	0.5
P (unitless)	1.0
k_0 (yr^{-1})	0.0
θ (mL water/cm^3 soil)	0.22
$t_{½}$ (yr)	2.49
K_d (m^3/kg)	120
R_t (event rainfall) (cm)	5.0
M_t (event snow melt) (cm)	0.0
CN (SCS Curve Number)	78.0
T_r (storm duration) (hrs)	6.0
VS_x (storm stream flow volume) (m^3)	8.63×10^4
Time (event) (yr)	2.74×10^{-3}
IR (cm/yr)	1.0

values used for these parameters in the three different scenarios (A, B and C) are given in Table 9-5. Sample calculations for each scenario follow.

9.5.1.1. SCENARIO A -- Tier 1 values for the receiving water concentration for Scenario A can be calculated directly from the input variables in Tables 9-4 and 9-5. The concentration of benzo(a)pyrene in the receiving water for long-term loading in Tier 1, Ci_{Lg1} (mg/L), can be calculated using Equation 9-1 and converting to the proper units:

$$Ci_{Lg1} = \frac{6.51\text{x}10^{-5}\ \text{kg/km}^3\text{-yr} \cdot (15\ \text{km}^2 + 1\ \text{km}^2)}{3.15\text{x}10^{7}\ \text{m}^3\text{/yr}} \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{Lg1} = 3.31\text{x}10^{-8}\ \text{mg/L}$$

The concentration in the receiving water for short-term loading in Tier 1, Ci_{Sh1} (mg/L), is obtained by replacing Vf_x with VS_x in Equation 9-1:

$$Ci_{Sh1} = \frac{6.51\text{x}10^{-5}\ \text{kg/km}^3\text{-yr} \cdot (15\ \text{km}^2 + 1\ \text{km}^2)}{8.63\text{x}10^{4}\ \text{m}^3\text{/yr}} \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{Sh1} = 1.21\text{x}10^{-5}\ \text{mg/L}$$

TABLE 9-5

Annual Fallout and Facility Life for Benzo(a)Pyrene

Scenario	Facility Life (timef, yr)	Fallout (Fa, kg/km^3-yr)
A	30	6.51×10^{-5}
B	60	9.17×10^{-4}
C	100	2.42×10^{-2}

For Tier 2 or 3 calculations, variables that are needed in subsequent equations must first be obtained. These are S (Equation 9-26), D_R (Equation 9-25), Q (Equation 9-28) and q_p (Equation 9-29):

$$S = 2.54 \cdot [(1000/78)-10] = 7.16$$

$$D_R = \frac{[5.0\ \text{cm} + 0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]^2}{5.0\ \text{cm} + 0\ \text{cm} + (0.8 \cdot 7.16\ \text{cm})} = 1.19\ \text{cm}$$

$$Q = 15\ \text{km}^2 \cdot 1.19\ \text{cm} = 17.8\ \text{km}^2\text{-cm}$$

$$q_p = \frac{15\ \text{km}^2 \cdot 1.19\ \text{cm} \cdot 5\ \text{cm}}{6.0\ \text{hr} \cdot [5.0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]} \cdot \frac{\text{hr}}{3.6\text{x}10^6\text{s}} \cdot \frac{\text{km}}{10^5\ \text{cm}} \cdot \frac{10^9\ \text{m}^3}{\text{km}^3}$$

$$q_p = 11.6\ \text{m}^3/\text{s}$$

The mass of eroded soil, X_s, is obtained from the Modified Universal Soil Loss Equation (Equation 9-27):

$$X_s = 2.04\text{x}10^6 \cdot (17.8 \cdot 11.55)^{0.56} \cdot (0.210)\ (0.179)\ (0.5)\ (1.0)$$
$$X_s = 7.56\text{x}10^5\ \text{kg}$$

Calculations for the long-term loading in Tier 2 or 3 can now be initiated. The rate of sediment loss to the receiving water is computed from the Universal Soil Loss Equation (Equation 9-3):

$$X_e = \frac{400}{\text{yr}} \cdot \frac{0.21\ \text{ton}}{\text{acre}} \cdot (0.179)\ (0.5)\ (1.0) \cdot \frac{907.18\ \text{kg}}{\text{ton}} \cdot \frac{\text{acre}}{4.047\text{x}10^{-3}\ \text{km}^2}$$

$$X_e = 1.68\text{x}10^6\ \text{kg/km-yr}$$

The first order loss rate, k_1 (Equation 9-6) can be calculated by adding the loss rates due to infiltration, k_{1I} (Equation 9-7), erosion, k_{1E} (Equation 9-8) and degradation, k_{1D} (Equation 9-9):

$$k_{1I} = \frac{1\ \text{cm/yr}}{0.22\ \text{mL/cm}^3 \cdot 1.0\ \text{cm} \cdot [1+(1500\ \text{mg/m}^3 \cdot 120\ \text{m}^3/\text{kg})/0.22\ \text{mL/cm}^3)]}$$

$$k_{1I} = 5.56\text{x}10^{-6}\ \text{yr}^{-1}$$

$$k_{1E} = \frac{1.685\text{x}10^6\ \text{kg/km-yr}}{1500\ \text{mg/m}^3 \cdot 1.0\ \text{cm}} \cdot \frac{120\ \text{m}^3/\text{kg} \cdot 1500\ \text{mg/m}^3}{0.22\ \text{mL/cm}^3 + (120\ \text{m}^3/\text{kg} \cdot 1500\ \text{mg/m}^3)}$$

$$k_{1E} = 0.112\ \text{yr}^{-1}$$

$$k_{1D} = \frac{\ln 2}{2.49} = 0.278\ \text{yr}^{-1}$$

$$k_1 = 5.56\text{x}10^{-6}\ \text{yr}^{-1} + 0.112\ \text{yr}^{-1} + 0.278\ \text{yr}^{-1} = 0.391\ \text{yr}^{-1}$$

The maximum contaminant mass per area of soil, Mm (Equation 9-4) is based on the assumption that the combustor has been in operation long enough to allow a steady state concentration to be reached:

$$Mm = 6.51\times10^{-5}\ kg/km^2\text{-}yr \cdot \frac{1 - \exp(-0.391yr^{-1} \cdot 30.0\ yr)}{0.391yr^{-1}}$$

$$Mm = 1.67\times10^{-4}\ kg/km^2$$

The contaminant also enters the water body by direct deposition (Equation 9-5):

$$Mw = 6.51\times10^{-5}\ kg/km^2\text{-}yr \cdot \frac{1\ km^2}{3.15\times10^{7}\ m^3/yr}$$

$$Mw = 2.07\times10^{-12}\ kg/m^3$$

For long-term loading, the receiving water concentration can now be calculated using Equation 9-2:

$$Ci_{Lg23} = \frac{1.68\times10^{6}\ kg/km\text{-}yr \cdot 15\ km^2 \cdot 1.67\times10^{-4}\ kg/km^2}{3.15\times10^{7}\ m^3/yr \cdot 1500\ kg/m^3 \cdot 1.00\ cm} \cdot \frac{10^5\ cm}{km} \cdot \frac{km^3}{10^6\ m^3} + 2.07\times10^{-12}\ kg/m^3 \cdot \frac{10^6\ mg}{kg} \cdot \frac{m^3}{10^3\ L}$$

$$Ci_{Lg23} = 1.10\times10^{-8}\ mg/L$$

For short-term loading, the Tier 2 and 3 model calculates the receiving water concentration after a storm event. It takes into account any repartitioning of the contaminant upon dilution. This is initiated by calculating the mean sediment concentration in the water body, C_s, Equation 9-17:

$$C_s = \frac{7.56\times10^{5}\ kg}{8.63\times10^{4}\ m^3/yr} = 8.77\ kg/m^3$$

The fraction that would be present in the dissolved, α_d (Equation 9-15), and adsorbed, α_a (Equation 9-16), state can now be determined:

$$\alpha_d = \frac{1}{1 + (120\ \text{m}^3/\text{kg} \cdot 8.77\ \text{kg/m}^3)} = 9.50\text{x}10^{-4}$$

$$\alpha_a = \frac{120\ \text{m}^3/\text{kg} \cdot 8.77\ \text{kg/m}^3}{1 + (120\ \text{m}^3/\text{kg} \cdot 8.77\ \text{kg/m}^3)} = 0.999$$

After a storm event, the amount of contaminant entering the water body will be partitioned between the adsorbed, Aa (Equation 9-23), and dissolved, Da (Equation 9-24), state:

$$Aa = [1 \div (1 + \frac{0.220\ \text{mL/cm}^3}{120\ \text{m}^3/\text{kg} \cdot 1500\ \text{kg/m}^3})] \cdot 1.67\text{x}10^{-4}\ \text{kg/km}^2$$

$$Aa = 4.44\text{x}10^{-5}\ \text{kg/km}^2$$

$$Da = [1 \div (1 + \frac{120\ \text{m}^3/\text{kg} \cdot 1500\ \text{kg/m}^3}{0.220\ \text{mL/cm}^3})] \cdot 1.67\text{x}10^{-4}\ \text{kg/km}^2$$

$$Da = 2.04\text{x}10^{-10}\ \text{kg/km}^2$$

The amount of chemical deposited directly onto the water body is a function of the deposition rate, area of the water body, and time (Equation 9-20):

$$P_{ft} = 6.51\text{x}10^{-5}\ \text{kg/km}^2\text{-yr} \cdot 2.74\text{x}10^{-3}\ \text{yr} \cdot 1.0\ \text{km}^2$$

$$P_{ft} = 1.78\text{x}10^{-7}\ \text{kg}$$

The mass of contaminant entering the water body from the storm runoff in the adsorbed form, P_{xt} (Equation 9-21), and the dissolved form, P_{qt} (Equation 9-22), is obtained as follows:

$$P_{xt} = \frac{7.56\text{x}10^5 \text{ kg}}{1500 \text{ mg/m}^3 \cdot 1.00 \text{ cm}} \cdot 4.44\text{x}10^{-5} \text{ kg/km}^2 \cdot \frac{10^5 \text{ cm}}{\text{km}} \cdot \frac{\text{km}^2}{10^9 \text{ m}^3}$$

$$P_{xt} = 2.24\text{x}10^{-6} \text{ kg}$$

$$P_{qt} = \frac{1.19 \text{ cm}}{(5.0 \text{ cm} + 0.00 \text{ cm})} \cdot 2.04\text{x}10^{-10} \text{ kg/km}^2 \cdot 15 \text{ km}^2$$

$$P_{qt} = 7.24\text{x}10^{-10} \text{ kg}$$

The total amount of the chemical in the dissolved state, $CHEM_d$ (Equation 9-13), and the adsorbed form, $CHEM_a$ (Equation 9-14), can be determined by first calculating the total amount of the contaminant entering the water body from all sources, TOTCHEM (Equation 9-19), and multiplying by the distribution factors:

$$\text{TOTCHEM} = 2.24\text{x}10^{-6} \text{ kg} + 7.24\text{x}10^{-10} \text{ kg} + 1.78\text{x}10^{-7} \text{ kg}$$

$$\text{TOTCHEM} = 2.42\text{x}10^{-6} \text{ kg}$$

$$CHEM_d = 2.42\text{x}10^{-6} \text{ kg} \cdot 9.50\text{x}10^{-4} \cdot \frac{10^6 \text{ mg}}{\text{kg}}$$

$$CHEM_d = 2.30\text{x}10^{-3} \text{ mg}$$

$$CHEM_a = 2.42\text{x}10^{-6} \text{ kg} \cdot 0.991 \cdot \frac{10^6 \text{ mg}}{\text{kg}}$$

$$CHEM_a = 2.42 \text{ mg}$$

The final concentrations for the short-term Tier 2 or 3 scenario can now be determined. $CHEM_w$, the dissolved concentration of the contaminant in the water body in question is obtained from the dissolved mass and the appropriate dilution volume (Equation 9-10):

$$CHEM_w = \frac{2.30\text{x}10^{-3}\ \text{mg}}{8.63\text{x}10^{4}\ \text{m}^3} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$CHEM_w = 2.66\text{x}10^{-11}\ \text{mg/L}$$

The mass of adsorbed contaminant per mass of sediment in the water body, $CHEM_{s1}$ (Equation 9-11):

$$CHEM_{s1} = \frac{2.42\ \text{mg}}{7.56\text{x}10^{5}\ \text{kg}}$$

$$CHEM_{s1} = 3.20\text{x}10^{-6}\ \text{mg/kg}$$

The mass of adsorbed contaminant per volume of water, $CHEM_{s2}$ (Equation 9-12):

$$CHEM_{s2} = \frac{2.42\ \text{mg}}{8.63\text{x}10^{4}\ \text{m}^3} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$CHEM_{s2} = 2.80\text{x}10^{-8}\ \text{mg/L}$$

9.5.1.2. SCENARIO B -- Tier 1 values for the receiving water concentration of benzo(a)pyrene for the B scenario can also be calculated directly from the input variables in Tables 9-4 and 9-5. The concentration of benzo(a)pyrene in the receiving water for long-term loading in Tier 1, Ci_{Lg1} (mg/L), can be calculated using Equation 9-1 and converting to the proper units:

$$Ci_{Lg1} = \frac{9.17\text{x}10^{-4}\ \text{kg/km}^3\text{-yr} \cdot (15\ \text{km}^2 + 1\ \text{km}^2)}{3.15\text{x}10^{7}\ \text{m}^3\text{/yr}} \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{Lg1} = 4.66\text{x}10^{-7}\ \text{mg/L}$$

The concentration in the receiving water for short-term loading in Tier 1, Ci_{Sh1} (mg/L), is obtained by replacing Vf_x with VS_x in Equation 9-1:

$$Ci_{Sh1} = \frac{9.17\text{x}10^{-4}\ \text{kg/km}^3\text{-yr} \cdot (15\ \text{km}^2 + 1\ \text{km}^2)}{8.63\text{x}10^{4}\ \text{m}^3\text{/yr}} \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{Sh1} = 1.70\text{x}10^{-4}\ \text{mg/L}$$

For Tier 2 or 3 calculations, variables that are needed in subsequent equations must first be obtained. These are S (Equation 9-26), D_R (Equation 9-25), Q (Equation 9-28) and q_p (Equation 9-29):

$$S = 2.54 \cdot [(1000/78)-10] = 7.16$$

$$D_R = \frac{[5.0\ \text{cm} + 0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]^2}{5.0\ \text{cm} + 0\ \text{cm} + (0.8 \cdot 7.16\ \text{cm})} = 1.19\ \text{cm}$$

$$Q = 15\ \text{km}^2 \cdot 1.19\ \text{cm} = 17.8\ \text{km}^2\text{-cm}$$

$$q_p = \frac{15\ \text{km}^2 \cdot 1.19\ \text{cm} \cdot 5\ \text{cm}}{6.0\ \text{hr} \cdot [5.0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]} \cdot \frac{\text{hr}}{3.6\text{x}10^6\ \text{s}} \cdot \frac{\text{km}}{10^5\ \text{cm}} \cdot \frac{10^9\ \text{m3}}{\text{km}^3}$$

$$q_p = 11.6\ \text{m}^3/\text{s}$$

The mass of eroded soil, X_s, is obtained from the Modified Universal Soil Loss Equation (Equation 9-27):

$$X_s = 2.04\text{x}10^6 \cdot (17.8 \cdot 11.55)^{0.56} \cdot (0.210)\ (0.179)\ (0.5)\ (1.0)$$

$$X_s = 7.56\text{x}10^5\ \text{kg}$$

Calculations for the long-term scenario can now be initiated. The rate of sediment loss to the receiving water is computed from the Universal Soil Loss Equation (Equation 9-3):

$$X_e = \frac{400}{\text{yr}} \cdot \frac{0.21\ \text{ton}}{\text{acre}} \cdot (0.179)\ (0.5)\ (1.0) \cdot \frac{907.18\ \text{kg}}{\text{ton}} \cdot \frac{\text{acre}}{4.047\text{x}10^{-3}\ \text{km}^2}$$

$$X_e = 1.68\text{x}10^6\ \text{kg/km-yr}$$

The first order loss rate, k_1 (Equation 9-6) can be calculated by adding the loss rates due to infiltration, k_{1I} (Equation 9-7), erosion, k_{1E} (Equation 9-8) and degradation, k_{1D} (Equation 9-9):

$$k_{1I} = \frac{1\ \text{cm/yr}}{0.22\ \text{mL/cm}^3 \cdot 1.0\ \text{cm} \cdot [1+(1500\ \text{mg/m}^3 \cdot 120\ \text{m}^3/\text{kg})/0.22\ \text{mL/cm}^3)]}$$

$$k_{1I} = 5.56\text{x}10^{-6}\ \text{yr}^{-1}$$

$$k_{1E} = \frac{1.685\text{x}10^{6}\ \text{kg/km-yr}}{1500\ \text{mg/m}^3 \cdot 1.0\ \text{cm}} \cdot \frac{120\ \text{m}^3/\text{kg} \cdot 1500\ \text{mg/m}^3}{0.22\ \text{mL/cm}^3 + (120\ \text{m}^3/\text{kg} \cdot 1500\ \text{mg/m}^3)}$$

$$k_{1E} = 0.112\ \text{yr}^{-1}$$

$$k_{1D} = \frac{\ln 2}{2.49} = 0.278\ \text{yr}^{-1}$$

$$k_1 = 5.56\text{x}10^{-6}\ \text{yr}^{-1} + 0.112\ \text{yr}^{-1} + 0.278\ \text{yr}^{-1} = 0.391\ \text{yr}^{-1}$$

The maximum contaminant mass per area of soil, Mm (Equation 9-4) is based on the assumption that the combustor has been in operation long enough to allow a steady state concentration to be reached:

$$Mm = 9.17\text{x}10^{-4}\ \text{kg/km}^2\text{-yr} \cdot \frac{1 - \exp(-0.391\ \text{yr}^{-1} \cdot 60.0\ \text{yr})}{0.391\ \text{yr}^{-1}}$$

$$Mm = 2.35\text{x}10^{-3}\ \text{kg/km}^2$$

The contaminant also enters the water body by direct deposition (Equation 9-5):

$$Mw = 9.17\text{x}10^{-4}\ \text{kg/km}^2\text{-yr} \cdot \frac{1\ \text{km}^2}{3.15\text{x}10^{7}\ \text{m}^3/\text{yr}}$$

$$Mw = 2.91\text{x}10^{-11}\ \text{kg/m}^3$$

For long-term loading, the receiving water concentration can now be calculated using Equation 9-2:

$$Ci_{Lg23} = \frac{1.68\mathrm{x}10^{6}\ \mathrm{kg/km\text{-}yr} \cdot 15\ \mathrm{km}^2 \cdot 2.35\mathrm{x}10^{-3}\ \mathrm{kg/km}^2}{3.15\mathrm{x}10^{9}\ \mathrm{m^3/yr} \cdot 1500\ \mathrm{kg/m^3} \cdot 1.00\ \mathrm{cm}} \cdot \frac{10^5\ \mathrm{cm}}{\mathrm{km}} \cdot \frac{\mathrm{km}^3}{10^6\ \mathrm{m}^3} + 2.91\mathrm{x}10^{-11}\ \mathrm{kg/m^3} \cdot \frac{10^6\ \mathrm{mg}}{\mathrm{kg}} \cdot \frac{\mathrm{m}^3}{10^3\ \mathrm{L}}$$

$$Ci_{Lg23} = 1.55\mathrm{x}10^{-7}\ \mathrm{mg/L}$$

For the short-term loading scenario, the Tier 2 and 3 model calculates the receiving water concentration after a storm event. It takes into account any repartitioning of the contaminant upon dilution. This is initiated by calculating the mean sediment concentration in the water body, C_s, Equation 9-17:

$$C_s = \frac{7.56\mathrm{x}10^{5}\ \mathrm{kg}}{8.63\mathrm{x}10^{4}\ \mathrm{m^3/yr}} = 8.77\ \mathrm{kg/m^3}$$

The fraction that would be present in the dissolved, α_d (Equation 9-15), and adsorbed, α_a (Equation 9-16), state can now be determined:

$$\alpha_d = \frac{1}{1 + (120\ \mathrm{m^3/kg} \cdot 8.77\ \mathrm{kg/m^3})} = 9.50\mathrm{x}10^{-4}$$

$$\alpha_a = \frac{120\ \mathrm{m^3/kg} \cdot 8.77\ \mathrm{kg/m^3}}{1 + (120\ \mathrm{m^3/kg} \cdot 8.77\ \mathrm{kg/m^3})} = 0.999$$

After a storm event, the amount of contaminant entering the water body will be partitioned between the adsorbed, Aa (Equation 9-23), and dissolved, Da (Equation 9-24), state:

$$Aa = [1/(\ 1 + \frac{0.220\ mL/cm^3}{120\ m^3/kg \cdot 1500\ kg/m^3}\ )] \cdot 2.35x10^{-3}\ kg/km^2$$

$$Aa = 6.26x10^{-4}\ kg/km^2$$

$$Da = [1/(\ 1 + \frac{120\ m^3/kg \cdot 1500\ kg/m^3}{0.220\ mL/cm^3}\ )] \cdot 2.35x10^{-3}\ kg/km^2$$

$$Da = 2.87x10^{-9}\ kg/km^2$$

The amount of chemical deposited directly onto the water body is a function of the deposition rate, area of the water body, and time (Equation 9-20):

$$P_{ft} = 9.17x10^{-4}\ kg/km^2\text{-}yr \cdot 2.74x10^{-3}\ yr \cdot 1.0\ km^2$$

$$P_{ft} = 2.51x10^{-6}\ kg$$

The mass of contaminant entering the water body from the storm runoff in the adsorbed form, P_{xt} (Equation 9-21), and the dissolved form, P_{qt} (Equation 9-22), is obtained as follows:

$$P_{xt} = \frac{7.56x10^5\ kg}{1500\ mg/m^3 \cdot 1.00\ cm} \cdot 6.26x10^{-4}\ kg/km^2 \cdot \frac{10^5\ cm}{km} \cdot \frac{km^2}{10^9\ m^3}$$

$$P_{xt} = 3.16x10^{-5}\ kg$$

$$P_{qt} = \frac{1.19\ cm}{(5.0\ cm + 0.00\ cm)} \cdot 2.87x10^{-9}\ kg/km^2 \cdot 15\ km^2$$

$$P_{qt} = 1.02x10^{-8}\ kg$$

The total amount of the chemical in the dissolved state, $CHEM_d$ (Equation 9-13), and the adsorbed form, $CHEM_a$ (Equation 9-14), can be determined by first calculating the total amount of the contaminant entering the water body from all sources, TOTCHEM (Equation 9-19), and multiplying by the distribution factors:

$$\text{TOTCHEM} = 3.16\text{x}10^{-5}\ \text{kg} + 1.02\text{x}10^{-8}\ \text{kg} + 2.51\text{x}10^{-6}\ \text{kg}$$

$$\text{TOTCHEM} = 3.41\text{x}10^{-5}\ \text{kg}$$

$$\text{CHEM}_d = 3.41\text{x}10^{-5}\ \text{kg} \cdot 9.50\text{x}10^{-4} \cdot \frac{10^6\ \text{mg}}{\text{kg}}$$

$$\text{CHEM}_d = 3.24\text{x}10^{-2}\ \text{mg}$$

$$\text{CHEM}_a = 3.41\text{x}10^{-5}\ \text{kg} \cdot 0.991 \cdot \frac{10^6\ \text{mg}}{\text{kg}}$$

$$\text{CHEM}_a = 34.0\ \text{mg}$$

The final concentrations for the short-term Tier 2 and Tier 3 scenario can now be determined. CHEM_w, the dissolved concentration of the contaminant in the water body in question is obtained from the dissolved mass and the appropriate dilution volume (Equation 9-10):

$$\text{CHEM}_w = \frac{3.24\text{x}10^{-2}\ \text{mg}}{8.63\text{x}10^{4}\ \text{m}^3} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$\text{CHEM}_w = 3.75\text{x}10^{-10}\ \text{mg/L}$$

The mass of adsorbed contaminant per mass of sediment in the water body, CHEM_{s1} (Equation 9-11):

$$\text{CHEM}_{s1} = \frac{34.0\ \text{mg}}{7.56\text{x}10^{5}\ \text{kg}}$$

$$\text{CHEM}_{s1} = 4.50\text{x}10^{-5}\ \text{mg/kg}$$

The mass of adsorbed contaminant per volume of water, CHEM_{s2} (Equation 9-12):

$$\text{CHEM}_{s2} = \frac{34.0\ \text{mg}}{8.63\text{x}10^{4}\ \text{m}^3} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$\text{CHEM}_{s2} = 3.95\text{x}10^{-7}\ \text{mg/L}$$

9.5.1.3. Scenario C -- Tier 1 values for the receiving water concentration for Scenario C can be calculated directly from the input variables in Tables 9-4 and 9-5. The concentration of benzo(a)pyrene in the receiving water for long-term loading in Tier 1, Ci_{Lg1} (mg/L), can be calculated using Equation 9-1 and converting to the proper units:

$$\text{Ci}_{Lg1} = \frac{2.42\text{x}10^{-2}\ \text{kg/km}^3\text{-yr} \cdot (15\ \text{km}^2 + 1\ \text{km}^2)}{3.15\text{x}10^{7}\ \text{m}^3/\text{yr}} \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$\text{Ci}_{Lg1} = 1.23\text{x}10^{-5}\ \text{mg/L}$$

The concentration in the receiving water for short-term loading in Tier 1, Ci_{Sh1} (mg/L), is obtained by replacing Vf_x with VS_x in Equation 9-1:

$$Ci_{Sh1} = \frac{2.42\text{x}10^{-2}\ \text{kg/km}^3\text{-yr} \cdot (15\ \text{km}^2 + 1\ \text{km}^2)}{8.63\text{x}10^4\ \text{m}^3/\text{yr}} \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{Sh1} = 4.49\text{x}10^{-3}\ \text{mg/L}$$

For Tier 2 or Tier 3 calculations, variables that are needed in subsequent equations must first be obtained. These are S (Equation 9-26), D_R (Equation 9-25), Q (Equation 9-28) and q_p (Equation 9-29):

$$S = 2.54 \cdot [(1000/78)-10] = 7.16$$

$$D_R = \frac{[5.0\ \text{cm} + 0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]^2}{5.0\ \text{cm} + 0\ \text{cm} + (0.8 \cdot 7.16\ \text{cm})} = 1.19\ \text{cm}$$

$$Q = 15\ \text{km}^2 \cdot 1.19\ \text{cm} = 17.8\ \text{km}^2\text{-cm}$$

$$q_p = \frac{15\ \text{km}^2 \cdot 1.19\ \text{cm} \cdot 5\ \text{cm}}{6.0\ \text{hr} \cdot [5.0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]} \cdot \frac{\text{hr}}{3.6\text{x}10^6\text{s}} \cdot \frac{\text{km}}{10^5\ \text{cm}} \cdot \frac{10^9\ \text{m}^3}{\text{km}^3}$$

$$q_p = 11.6\ \text{m}^3/\text{s}$$

The mass of eroded soil, X_s, is obtained from the Modified Universal Soil Loss Equation (Equation 9-27):

$$X_s = 2.04\text{x}10^6 \cdot (17.8 \cdot 11.55)^{0.56} \cdot (0.210)\ (0.179)\ (0.5)\ (1.0)$$

$$X_s = 7.56\text{x}10^5\ \text{kg}$$

Calculations for the long-term scenario can now be initiated. The rate of sediment loss to the receiving water is computed from the Universal Soil Loss Equation (Equation 9-3):

$$X_e = \frac{400}{\text{yr}} \cdot \frac{0.21 \text{ ton}}{\text{acre}} \cdot (0.179)\ (0.5)\ (1.0) \cdot \frac{907.18 \text{ kg}}{\text{ton}} \cdot \frac{\text{acre}}{4.047\text{x}10^{-3}\text{km}^2}$$

$$X_e = 1.68\text{x}10^6 \text{ kg/km-yr}$$

The first order loss rate, k_1 (Equation 9-6) can be calculated by adding the loss rates due to infiltration, k_{1I} (Equation 9-7), erosion, k_{1E} (Equation 9-8) and degradation, k_{1D} (Equation 9-9):

$$k_{1I} = \frac{1 \text{ cm/yr}}{0.22 \text{ mL/cm}^3 \cdot 1.0 \text{ cm} \cdot [1 + (1500 \text{ mg/m}^3 \cdot 120 \text{ m}^3\text{/kg})/0.22 \text{ mL/cm}^3)]}$$

$$k_{1I} = 5.56\text{x}10^{-6} \text{ yr}^{-1}$$

$$k_{1E} = \frac{1.685\text{x}10^6 \text{ kg/km-yr}}{1500 \text{ mg/m}^3 \cdot 1.0 \text{ cm}} \cdot \frac{120 \text{ m}^3\text{/kg} \cdot 1500 \text{ mg/m}^3}{0.22 \text{ mL/cm}^3 + (120 \text{ m}^3\text{/kg} \cdot 1500 \text{ mg/m}^3)}$$

$$k_{1E} = 0.112 \text{ yr}^{-1}$$

$$k_{1D} = \frac{\ln 2}{2.49} = 0.278 \text{ yr}^{-1}$$

$$k_1 = 5.56\text{x}10^{-6} \text{ yr}^{-1} + 0.112 \text{ yr}^{-1} + 0.278 \text{ yr}^{-1} = 0.391 \text{ yr}^{-1}$$

The maximum contaminant mass per area of soil, Mm (Equation 9-4) is based on the assumption that the combustor has been in operation long enough to allow a steady state concentration to be reached:

$$Mm = 2.42\text{x}10^{-2}\ \text{kg/km}^2\text{-yr} \cdot \frac{1 - \exp(-0.391\ \text{yr}^{-1} \cdot 100\ \text{yr})}{0.391\ \text{yr}^{-1}}$$

$$Mm = 0.0619\ \text{kg/km}^2$$

The contaminant also enters the water body by direct deposition (Equation 9-5):

$$Mw = 2.42\text{x}10^{-2}\ \text{kg/km}^2\text{-yr} \cdot \frac{1\ \text{km}^2}{3.15\text{x}10^{7}\ \text{m}^3\text{/yr}}$$

$$Mw = 7.68\text{x}10^{-10}\ \text{kg/m}^3$$

For the long-term loading scenario, the receiving water concentration can now be calculated using Equation 9-2:

$$Ci_{Lg23} = \frac{1.68\text{x}10^{6}\ \text{kg/km-yr} \cdot 15\ \text{km}^2 \cdot 0.0619\ \text{kg/km}^2}{3.15\text{x}10^{7}\ \text{m}^3\text{/yr} \cdot 1500\ \text{kg/m}^3 \cdot 1.00\ \text{cm}} \cdot \frac{10^5\ \text{cm}}{\text{km}} \cdot \frac{\text{km}^3}{10^6\ \text{m}^3}$$

$$+ 7.68\text{x}10^{-10}\ \text{kg/m}^3 \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{Lg23} = 4.08\text{x}10^{-6}\ \text{mg/L}$$

For the short-term loading scenario, the Tier 2 or 3 model calculates the receiving water concentration after a storm event. It takes into account any repartitioning of the contaminant upon dilution. This is initiated by calculating the mean sediment concentration in the water body, C_s, Equation 9-17:

$$C_s = \frac{7.56\text{x}10^5 \text{ kg}}{8.63\text{x}10^4 \text{ m}^3/\text{yr}} = 8.77 \text{ kg/m}^3$$

The fraction that would be present in the dissolved, α_d (Equation 9-15), and adsorbed, α_a (Equation 9-16), state can now be determined:

$$\alpha_d = \frac{1}{1 + (120 \text{ m}^3/\text{kg} \cdot 8.77 \text{ kg/m}^3)} = 9.50\text{x}10^{-4}$$

$$\alpha_a = \frac{120 \text{ m}^3/\text{kg} \cdot 8.77 \text{ kg/m}^3}{1 + (120 \text{ m}^3/\text{kg} \cdot 8.77 \text{ kg/m}^3)} = 0.999$$

After a storm event, the amount of contaminant entering the water body will be partitioned between the adsorbed, Aa (Equation 9-23), and dissolved, Da (Equation 9-24), state:

$$Aa = [1/(1 + \frac{0.220 \text{ mL/cm}^3}{120 \text{ m}^3/\text{kg} \cdot 1500 \text{ kg/m}^3})] \cdot 0.0619 \text{ kg/km}^2$$

$$Aa = 0.0165 \text{ kg/km}^2$$

$$Da = [\ 1/(\ 1 + \frac{120 \text{ m}^3/\text{kg} \cdot 1500 \text{ kg/m}^3}{0.220 \text{ mL/cm}^3}\ )\ ] \cdot 0.0619 \text{ kg/km}^2$$

$$Da = 7.57\text{x}10^{-8} \text{ kg/km}^2$$

The amount of chemical deposited directly onto the water body is a function of the deposition rate, area of the water body, and time (Equation 9-20):

P_{ft} = 0.0242 kg/km^2-yr · 2.74x10^{-3} yr · 1.0 km^2

P_{ft} = 6.63x10^{-5} kg

The mass of contaminant entering the water body from the storm runoff in the adsorbed form, P_{xt} (Equation 9-21), and the dissolved form, P_{qt} (Equation 9-22), is obtained as follows:

$$P_{xt} = \frac{7.56\text{x}10^{5}\ \text{kg}}{1500\ \text{mg/m}^3 \cdot 1.00\ \text{cm}} \cdot 0.0165\ \text{kg/km}^2 \cdot \frac{10^5\ \text{cm}}{\text{km}} \cdot \frac{\text{km}^2}{10^9\ \text{m}^3}$$

P_{xt} = 8.33x0^{-4} kg

$$P_{qt} = \frac{1.19\ \text{cm}}{(5.0\ \text{cm} + 0.00\ \text{cm})} \cdot 7.57\text{x}10^{-8}\ \text{kg/km}^2 \cdot 15\ \text{km}^2$$

P_{qt} = 2.69x10^{-7} kg

The total amount of the chemical in the dissolved state, CHEM$_d$ (Equation 9-13), and the adsorbed form, CHEM$_a$ (Equation 9-14), can be determined by first calculating the total amount of the contaminant entering the water body from all sources, TOTCHEM (Equation 9-19), and multiplying by the distribution factors:

TOTCHEM = 8.33x10^{-4} kg + 2.69x10^{-6} kg + 6.636x10^{-5} kg

TOTCHEM = 8.99x10^{-4} kg

$$CHEM_d = 8.99x10^{-4}\ kg \cdot 9.50x10^{-4} \cdot \frac{10^6\ mg}{kg}$$

$$CHEM_d = 0.854\ mg$$

$$CHEM_a = 8.99x10^{-4}\ kg \cdot 0.991 \cdot \frac{10^6\ mg}{kg}$$

$$CHEM_a = 899\ mg$$

The final concentrations for the short-term Tier 2 or 3 scenario can now be determined. $CHEM_w$, the dissolved concentration of the contaminant in the water body in question is obtained from the dissolved mass and the appropriate dilution volume (Equation 9-10):

$$CHEM_w = \frac{0.854\ mg}{8.63x10^4\ m^3} \cdot \frac{m^3}{10^3\ L}$$

$$CHEM_w = 9.90x10^{-9}\ mg/L$$

The mass of adsorbed contaminant per mass of sediment in the water body, $CHEM_{s1}$ (Equation 9-11):

$$CHEM_{s1} = \frac{899\ mg}{7.56x10^5\ kg}$$

$$CHEM_{s1} = 1.19x10^{-3}\ mg/kg$$

The mass of adsorbed contaminant per volume of water, $CHEM_{s2}$ (Equation 9-12):

$$\mathrm{CHEM}_{s2} = \frac{899\ \mathrm{mg}}{8.63\times10^{4}\ \mathrm{m}^{3}} \cdot \frac{\mathrm{m}^{3}}{10^{3}\ \mathrm{L}}$$

$$\mathrm{CHEM}_{s2} = 1.04\times10^{-5}\ \mathrm{mg/L}$$

The final results for the concentration of the contaminant in the water body for the A, B and C scenarios are summarized in Table 9-6.

9.5.2. Cadmium. The values chosen for cadmium for the input variables of the Surface Runoff Model are given in Table 9-7. The concentration of a contaminant in the water is a function of its deposition rate, and the lifetime of the facility. Annual fallout (Fa) and facility lifetimes (timef) are given in Table 9-8 for Scenarios A, B and C. Sample calculations for each scenario follow.

9.5.2.1. SCENARIO A -- Tier 1 values for the receiving water concentration can be calculated directly from the input variables in Tables 9-7 and 9-8. The concentration of cadmium in the receiving water for long-term loading in Tier 1, Ci_{Lg1} (mg/L), can be calculated using Equation 9-1 and converting to the proper units:

$$\mathrm{Ci}_{Lg1} = \frac{8.94\times10^{-3}\ \mathrm{kg/km3\text{-}yr} \cdot (15\ \mathrm{km}^{2} + 1\ \mathrm{km}^{2})}{3.15\times10^{7}\ \mathrm{m}^{3}/\mathrm{yr}} \cdot \frac{10^{6}\ \mathrm{mg}}{\mathrm{kg}} \cdot \frac{\mathrm{m}^{3}}{10^{3}\ \mathrm{L}}$$

$$\mathrm{Ci}_{Lg1} = 4.54\times10^{-6}\ \mathrm{mg/L}$$

Table 9-6

Benzo(a)Pyrene Concentration in Receiving Water for the Three Different Scenarios

	A	B	C
Fallout (kg/m^3-yr)	6.51×10^{-5}	9.17×10^{-4}	2.42×10^{-2}
Facility lifetime (yr)	30	60	100
Ci_{Lg1} (mg/L)	3.31×10^{-8}	4.66×10^{-7}	1.23×10^{-5}
Ci_{Sh1} (mg/L)	1.21×10^{-5}	1.70×10^{-4}	4.49×10^{-3}
Ci_{Lg23} (mg/L)	1.10×10^{-8}	1.55×10^{-7}	4.08×10^{-6}
$CHEM_w$ (mg/L)	2.66×10^{-11}	3.75×10^{-10}	9.90×10^{-9}
$CHEM_{s1}$ (mg/kg)	3.20×10^{-6}	4.50×10^{-5}	1.19×10^{-3}
$CHEM_{s2}$ (mg/L)	2.80×10^{-8}	3.95×10^{-7}	1.04×10^{-5}

Table 9-7

Input Variables for Determination of Cadmium Water Concentrations

Variable	Value
WA_L (watershed area) (km^2)	15.0
WA_W (water body area) (km^2)	1.0
Vf_x (m^3/yr)	3.15×10^7
BD (bulk soil density) (mg/m^3)	1500
Z (soil till depth) (cm)	1.0
R (yr^{-1})	400.0
K (tons/acre)	0.21
LS (unitless)	0.179
C (unitless)	0.5
P (unitless)	1.0
k_0 (yr^{-1})	0.0
θ (mL water/cm^3 soil)	0.22
$t_{1/2}$ (yr)	100
K_d (m^3/kg)	0.500
R_t (event rainfall) (cm)	5.0
M_t (event snow melt) (cm)	0.0
CN (SCS Curve Number)	78.0
T_r (storm duration) (hrs)	6.0
VS_x (storm stream flow volume) (m^3)	8.63×10^4
Time (event) (yr)(m^3)	2.74×10^{-3}
IR (cm/yr)	1.0

TABLE 9-8

Annual Fallout and Facility Life for Cadmium

Scenario	Facility Life (timef, yr)	Fallout (Fa, kg/km^3-yr)
A	30	8.94×10^{-3}
B	60	3.76×10^{-1}
C	100	11.9

The concentration in the receiving water for short-term loading in Tier 1, Ci_{sh1} (mg/L), is obtained by replacing Vf_x with VS_x in Equation 9-1:

$$Ci_{sh1} = \frac{8.94\text{x}10^{-3}\ \text{kg/km}^3\text{-yr} \cdot (15\ \text{km}^2 + 1\ \text{km}^2)}{8.63\text{x}10^4\ \text{m}^3/\text{yr}} \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{sh1} = 1.66\text{x}10^{-3}\ \text{mg/L}$$

For Tier 2 and 3 calculations, variables that are needed in subsequent equations must first be obtained. These are S (Equation 9-26), D_R (Equation 9-25), Q (Equation 9-28) and q_p (Equation 9-29):

$$S = 2.54 \cdot [(1000/78)-10] = 7.16$$

$$D_R = \frac{[5.0\ \text{cm} + 0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]^2}{5.0\ \text{cm} + 0\ \text{cm} + (0.8 \cdot 7.16\ \text{cm})} = 1.19\ \text{cm}$$

$$Q = 15\ \text{km}^2 \cdot 1.19\ \text{cm} = 17.8\ \text{km}^2\text{-cm}$$

$$q_p = \frac{15\ \text{km}^2 \cdot 1.19\ \text{cm} \cdot 5\ \text{cm}}{6.0\ \text{hr} \cdot [5.0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]} \cdot \frac{\text{hr}}{3.6\text{x}10^6\ \text{s}} \cdot \frac{\text{km}}{10^5\ \text{cm}} \cdot \frac{10^9\ \text{m}^3}{\text{km}^3}$$

$$q_p = 11.6\ \text{m}^3/\text{s}$$

The mass of eroded soil, X_s, is obtained from the Modified Universal Soil Loss Equation (Equation 9-27):

$$X_s = 2.04\text{x}10^6 \cdot (17.8 \cdot 11.55)^{0.56} \cdot (0.210)\ (0.179)\ (0.5)\ (1.0)$$

$$X_s = 7.56\text{x}10^5 \text{ kg}$$

Calculations for the long-term scenario can now be initiated. The rate of sediment loss to the receiving water is computed from the Universal Soil Loss Equation (Equation 9-3):

$$X_e = \frac{400}{\text{yr}} \cdot \frac{0.21 \text{ ton}}{\text{acre}} \cdot (0.179)\ (0.5)\ (1.0) \cdot \frac{907.18 \text{ kg}}{\text{ton}} \cdot \frac{\text{acre}}{4.047\text{x}10^{-3} \text{ km}^2}$$

$$X_e = 1.68\text{x}10^6 \text{ kg/km-yr}$$

The first order loss rate, k_1 (Equation 9-6) can be calculated by adding the loss rates due to infiltration, k_{1I} (Equation 9-7), erosion, k_{1E} (Equation 9-8) and degradation, k_{1D} (Equation 9-9):

$$k_{1I} = \frac{1 \text{ cm/yr}}{0.22 \text{ mL/cm}^3 \cdot 1.0 \text{ cm} \cdot [1 + (1500 \text{ mg/m}^3 \cdot 0.500 \text{ m}^3\text{/kg}) \div 0.22 \text{ mL/cm}^3)]}$$

$$k_{1I} = 1.33\text{x}10^{-3} \text{ yr}^{-1}$$

$$k_{1E} = \frac{1.68\text{x}10^6 \text{ kg/km-yr}}{1500 \text{ mg/m}^3 \cdot 1.0 \text{ cm}} \cdot \frac{0.500 \text{ m}^3\text{/kg} \cdot 1500 \text{ mg/m}^3}{0.22 \text{ mL/cm}^3 + (0.500 \text{ m}^3\text{/kg} \cdot 1500 \text{ mg/m}^3)}$$

$$k_{1E} = 0.112 \text{ yr}^{-1}$$

$$k_{1D} = \frac{\ln 2}{100} = 6.93\text{x}10^{-3} \text{ yr}^{-1}$$

$$k_1 = 1.33\text{x}10^{-3}\ \text{yr}^{-1} + 0.112\ \text{yr}^{-1} + 6.93\text{x}10^{-3}\ \text{yr}^{-1} = 0.121\ \text{yr}^{-1}$$

The maximum contaminant mass per area of soil, Mm (Equation 9-4) is based on the assumption that the combustor has been in operation long enough to allow a steady state concentration to be reached:

$$Mm = 8.94\text{x}10^{-3}\ \text{kg/km}^2\text{-yr} \cdot \frac{1 - \exp(-0.121\ \text{yr}^{-1} \cdot 30.0\ \text{yr})}{0.121\ \text{yr}^{-1}}$$

$$Mm = 7.21\text{x}10^{-2}\ \text{kg/km}^2$$

The contaminant also enters the water body by direct deposition (Equation 9-5):

$$Mw = 8.94\text{x}10^{-3}\ \text{kg/km}^2\text{-yr} \cdot \frac{1\ \text{km}^2}{3.15\text{x}10^{7}\ \text{m}^3/\text{yr}}$$

$$Mw = 2.84\text{x}10^{-10}\ \text{kg/m}^3$$

For long-term loading, the receiving water concentration can now be calculated using Equation 9-2:

$$Ci_{Lg23} = \frac{1.68\text{x}10^{8}\ \text{kg/km-yr} \cdot 15\ \text{km}^2 \cdot 93.7\ \text{kg/km}^2}{3.15\text{x}10^{7}\ \text{m}^3/\text{yr} \cdot 1500\ \text{kg/m}^3 \cdot 1.00\ \text{cm}} \cdot \frac{10^5\ \text{cm}}{\text{km}} \cdot \frac{\text{km}^3}{10^6\ \text{m}^3}$$
$$+ 3.78\text{x}10^{-7}\ \text{kg/m}^3 \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{Lg23} = 4.14\text{x}10^{-6}\ \text{mg/L}$$

For the short-term loading scenario, the Tier 2 and Tier 3 model calculates the receiving water concentration after a storm event. It takes into account any repartitioning of the contaminant upon dilution. This is initiated by calculating the mean sediment concentration in the water body, C_s, Equation 9-17:

$$C_s = \frac{7.56\text{x}10^5 \text{ kg}}{8.63\text{x}10^4 \text{ m}^3/\text{yr}} = 8.77 \text{ kg/m}^3$$

The fraction that would be present in the dissolved, α_d (Equation 9-15), and adsorbed, α_a (Equation 9-16), state can now be determined:

$$\alpha_d = \frac{1}{1 + (0.500 \text{ m}^3/\text{kg} \cdot 8.77 \text{ kg/m}^3)} = 0.186$$

$$\alpha_a = \frac{0.500 \text{ m}^3/\text{kg} \cdot 8.77 \text{ kg/m}^3}{1 + (0.500 \text{ m}^3/\text{kg} \cdot 8.77 \text{ kg/m}^3)} = 0.814$$

After a storm event, the amount of contaminant entering the water body will be partitioned between the adsorbed, Aa (Equation 9-23), and dissolved, Da (Equation 9-24), state:

$$\text{Aa} = [1 \div (1 + \frac{0.220 \text{ mL/cm}^3}{0.500 \text{ m}^3/\text{kg} \cdot 1500 \text{ kg/m}^3})] \cdot 7.21\text{x}10^{-2} \text{ kg/km}^2$$

$$\text{Aa} = 1.09\text{x}10^{-4} \text{ kg/km}^2$$

$$\text{Da} = [1 \div (1 + \frac{0.500 \text{ m}^3/\text{kg} \cdot 1500 \text{ kg/m}^3}{0.220 \text{ mL/cm}^3})] \cdot 7.21\text{x}10^{-2} \text{ kg/km}^2$$

$$\text{Da} = 2.12\text{x}10^{-5} \text{ kg/km}^2$$

The amount of chemical deposited directly onto the water body is a function of the annual fallout, area of the water body, and time (Equation 9-20):

$$P_{ft} = 8.94\text{x}10^{-3}\ \text{kg/km}^2\text{-yr}\cdot 2.74\text{x}10^{-3}\ \text{yr} \cdot 1.0\ \text{km}^2$$

$$P_{ft} = 2.45\text{x}10^{-5}\ \text{kg}$$

The mass of contaminant entering the water body from the storm runoff in the adsorbed form, P_{xt} (Equation 9-21), and the dissolved form, P_{qt} (Equation 9-22), is obtained as follows:

$$P_{xt} = \frac{7.56\text{x}10^{5}\ \text{kg}}{1500\ \text{mg/m}^3 \cdot 1.00\ \text{cm}} \cdot 1.09\text{x}10^{-4}\ \text{kg/km}^2 \cdot \frac{10^5\ \text{cm}}{\text{km}} \cdot \frac{\text{km}^2}{10^9\ \text{m}^3}$$

$$P_{xt} = 5.50\text{x}10^{-6}\ \text{kg}$$

$$P_{qt} = \frac{1.19\ \text{cm}}{(5.0\ \text{cm} + 0.00\ \text{cm})} \cdot 2.12\text{x}10^{-5}\ \text{kg/km}^2 \cdot 15\ \text{km}^2$$

$$P_{qt} = 7.52\text{x}10^{-5}\ \text{kg}$$

The total amount of the chemical in the dissolved state, $CHEM_d$ (Equation 9-13), and the adsorbed form, $CHEM_a$ (Equation 9-14), can be determined by first calculating the total amount of the contaminant entering the water body from all sources, TOTCHEM (Equation 9-19), and multiplying by the distribution factors:

$$\text{TOTCHEM} = 5.50\text{x}10^{-6}\ \text{kg} + 7.52\text{x}10^{-5}\ \text{kg} + 2.45\text{x}10^{-5}\ \text{kg}$$

$$\text{TOTCHEM} = 1.05\text{x}10^{-4}\ \text{kg}$$

$$\mathrm{CHEM}_d = 1.05\mathrm{x}10^{-4}\ \mathrm{kg} \cdot 0.186 \cdot \frac{10^6\ \mathrm{mg}}{\mathrm{kg}}$$

$$\mathrm{CHEM}_d = 19.6\ \mathrm{mg}$$

$$\mathrm{CHEM}_a = 1.05\mathrm{x}10^{-4}\ \mathrm{kg} \cdot 0.565 \cdot \frac{10^6\ \mathrm{mg}}{\mathrm{kg}}$$

$$\mathrm{CHEM}_a = 85.7\ \mathrm{mg}$$

The final concentrations for the short-term Tier 2 or 3 scenario can now be determined. CHEM_w, the dissolved concentration of the contaminant in the water body in question is obtained from the dissolved mass and the appropriate dilution volume (Equation 9-10):

$$\mathrm{CHEM}_w = \frac{19.6\ \mathrm{mg}}{8.63\mathrm{x}10^4\ \mathrm{m}^3} \cdot \frac{\mathrm{m}^3}{10^3\ \mathrm{L}}$$

$$\mathrm{CHEM}_w = 2.26\mathrm{x}10^{-7}\ \mathrm{mg/L}$$

The mass of adsorbed contaminant per mass of sediment in the water body, CHEM_{s1} (Equation 9-11):

$$\mathrm{CHEM}_{s1} = \frac{85.7\ \mathrm{mg}}{7.56\mathrm{x}10^5\ \mathrm{kg}}$$

$$\mathrm{CHEM}_{s1} = 1.13\mathrm{x}10^{-4}\ \mathrm{mg/kg}$$

The mass of adsorbed contaminant per volume of water, CHEM_{s2} (Equation 9-12):

$$\text{CHEM}_{s2} = \frac{85.7\text{ mg}}{8.63\text{x}10^{4}\text{ m}^{3}} \cdot \frac{\text{m}^{3}}{10^{3}\text{ L}}$$

$$\text{CHEM}_{s2} = 9.93\text{x}10^{-7}\text{ mg/L}$$

9.5.2.2. SCENARIO B -- For Scenario B, the lifetime of the facility (timef) changes to 60 years, and the deposition rate of cadmium (Fa) is $3.76\text{x}10^{-1}$ kg/km^2-yr. The calculations of the cadmium concentration for this scenario follow.

The concentration of cadmium in the receiving water for long-term loading in Tier 1, Ci_{Lg1} (mg/L), can be calculated using Equation 9-1 and converting to the proper units:

$$\text{Ci}_{Lg1} = \frac{3.76\text{x}10^{-1}\text{ kg/km}^{3}\text{-yr} \cdot (15\text{ km}^{2} + 1\text{ km}^{2})}{3.15\text{x}10^{7}\text{ m}^{3}\text{/yr}} \cdot \frac{10^{6}\text{ mg}}{\text{kg}} \cdot \frac{\text{m}^{3}}{10^{3}\text{ L}}$$

$$\text{Ci}_{Lg1} = 1.91\text{x}10^{-4}\text{ mg/L}$$

The concentration in the receiving water for short-term loading in Tier 1, Ci_{Sh1} (mg/L), is obtained by replacing Vf_x with VS_x in Equation 9-1:

$$\text{Ci}_{Sh1} = \frac{3.76\text{x}10^{-1}\text{ kg/km}^{3}\text{-yr} \cdot (15\text{ km}^{2} + 1\text{ km}^{2})}{8.63\text{x}10^{4}\text{ m}^{3}\text{/yr}} \cdot \frac{10^{6}\text{ mg}}{\text{kg}} \cdot \frac{\text{m}^{3}}{10^{3}\text{ L}}$$

$$\text{Ci}_{Sh1} = 6.97\text{x}10^{-2}\text{ mg/L}$$

For Tier 2 and 3 calculations, variables that are needed in subsequent equations must first be obtained. These are S (Equation 9-26), D_R (Equation 9-25), Q (Equation 9-28) and q_p (Equation 9-29):

$$S = 2.54 \cdot [(1000/78)-10] = 7.16$$

$$D_R = \frac{[5.0\ \text{cm} + 0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]^2}{5.0\ \text{cm} + 0\ \text{cm} + (0.8 \cdot 7.16\ \text{cm})} = 1.19\ \text{cm}$$

$$Q = 15\ \text{km}^2 \cdot 1.19\ \text{cm} = 17.8\ \text{km}^2\text{-cm}$$

$$q_p = \frac{15\ \text{km}^2 \cdot 1.19\ \text{cm} \cdot 5\ \text{cm}}{6.0\ \text{hr} \cdot [5.0\ \text{cm} - (0.2 \cdot 7.16\ \text{cm})]} \cdot \frac{\text{hr}}{3.6\text{x}10^6\ \text{s}} \cdot \frac{\text{km}}{10^5\ \text{cm}} \cdot \frac{10^9\ \text{m}^3}{\text{km}^3}$$

$$q_p = 11.6\ \text{m}^3/\text{s}$$

The mass of eroded soil, X_s, is obtained from the Modified Universal Soil Loss Equation (Equation 9-27):

$$X_s = 2.04\text{x}10^6 \cdot (17.8 \cdot 11.55)^{0.56} \cdot (0.210)\ (0.179)\ (0.5)\ (1.0)$$

$$X_s = 7.56\text{x}10^5\ \text{kg}$$

Calculations for the long-term scenario can now be initiated. The rate of sediment loss to the receiving water is computed from the Universal Soil Loss Equation (Equation 9-3):

$$X_e = \frac{400}{yr} \cdot \frac{0.21\ ton}{acre} \cdot (0.179)\ (0.5)\ (1.0) \cdot \frac{907.18\ kg}{ton} \cdot \frac{acre}{4.047\times10^{-3}\ km^2}$$

$$X_e = 1.68\times10^{6}\ kg/km\text{-}yr$$

The first order loss rate, k_1 (Equation 9-6) can be calculated by adding the loss rates due to infiltration, k_{1I} (Equation 9-7), erosion, k_{1E} (Equation 9-8) and degradation, k_{1D} (Equation 9-9):

$$k_{1I} = \frac{1\ cm/yr}{0.22\ mL/cm^3 \cdot 1.0\ cm \cdot [1 + (1500\ mg/m^3 \cdot 0.500\ m^3/kg) \div 0.22\ mL/cm^3)]}$$

$$k_{1I} = 1.33\times10^{-3}\ yr^{-1}$$

$$k_{1E} = \frac{1.68\times10^{6}\ kg/km\text{-}yr}{1500\ mg/m^3 \cdot 1.0\ cm} \cdot \frac{0.500\ m^3/kg \cdot 1500\ mg/m^3}{0.22\ mL/cm^3 + (0.500\ m^3/kg \cdot 1500\ mg/m^3)}$$

$$k_{1E} = 0.112\ yr^{-1}$$

$$k_{1D} = \frac{\ln 2}{100} = 6.93\times10^{-3}\ yr^{-1}$$

$$k_1 = 4.50\times10^{-3}\ yr^{-1} + 0.112\ yr^{-1} + 6.93\times10^{-3}\ yr^{-1} = 0.121\ yr^{-1}$$

The maximum contaminant mass per area of soil, Mm (Equation 9-4) is based on the assumption that the combustor has been in operation long enough to allow a steady state concentration to be reached:

$$Mm = 3.76\text{x}10^{-1}\ \text{kg/km}^2\text{-yr} \cdot \frac{1 - \exp(-0.121\ \text{yr}^{-1} \cdot 60\ \text{yr})}{0.121\ \text{yr}^{-1}}$$

$$Mm = 3.11\ \text{kg/km}^2$$

The contaminant also enters the water body by direct deposition (Equation 9-5):

$$Mw = 3.76\text{x}10^{-1}\ \text{kg/km}^2\text{-yr} \cdot \frac{1\ \text{km}^2}{3.15\text{x}10^{7}\ \text{m}^3\text{/yr}}$$

$$Mw = 1.19\text{x}10^{-8}\ \text{kg/m}^3$$

For the long-term loading scenario, the receiving water concentration can now be calculated using Equation 9-2:

$$Ci_{Lg23} = \frac{1.68\text{x}10^{8}\ \text{kg/km-yr} \cdot 15\ \text{km}^2 \cdot 3.11\ \text{kg/km}^2}{3.15\text{x}10^{7}\ \text{m}^3\text{/yr} \cdot 1500\ \text{kg/m}^3 \cdot 1.00\ \text{cm}} \cdot \frac{10^5\ \text{cm}}{\text{km}} \cdot \frac{\text{km}^3}{10^6\ \text{m}^3}$$
$$+\ 1.19\text{x}10^{-8}\ \text{kg/m}^3 \cdot \frac{10^6\ \text{mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3\ \text{L}}$$

$$Ci_{Lg23} = 1.79\text{x}10^{-4}\ \text{mg/L}$$

For the short-term loading scenario, the Tier 2 and Tier 3 model calculates the receiving water concentration after a storm event. It takes into account any repartitioning of the contaminant upon dilution. This is initiated by calculating the mean sediment concentration in the water body, C_s, Equation 9-17:

$$C_s = \frac{7.56\text{x}10^5\ \text{kg}}{8.63\text{x}10^4\ \text{m}^3\text{/yr}} = 8.77\ \text{kg/m}^3$$

The fraction that would be present in the dissolved, α_d (Equation 9-15), and adsorbed, α_a (Equation 9-16), state can now be determined:

$$\alpha_d = \frac{1}{1 + (0.500\ m^3/kg \cdot 8.77\ kg/m^3)} = 0.186$$

$$\alpha_a = \frac{0.500\ m^3/kg \cdot 8.77\ kg/m^3}{1 + (0.500\ m^3/kg \cdot 8.77\ kg/m^3)} = 0.814$$

After a storm event, the amount of contaminant entering the water body will be partitioned between the adsorbed, Aa (Equation 9-23), and dissolved, Da (Equation 9-24), state:

$$Aa = [1 \div (1 + \frac{0.220\ mL/cm^3}{0.500\ m^3/kg \cdot 1500\ kg/m^3})] \cdot 7.21x10^{-2}\ kg/km^2$$

$$Aa = 4.71x10^{-3}\ kg/km^2$$

$$Da = [1 \div (1 + \frac{0.500\ m^3/kg \cdot 1500\ kg/m^3}{0.220\ mL/cm^3})] \cdot 7.21x10^{-2}\ kg/km^2$$

$$Da = 9.13x10^{-4}\ kg/km^2$$

The amount of chemical deposited directly onto the water body is a function of the deposition rate, area of the water body, and time (Equation 9-20):

$$P_{ft} = 3.76x10^{-1}\ kg/km^2\text{-}yr \cdot 2.74x10^{-3}\ yr \cdot 1.0\ km^2$$

$$P_{ft} = 1.03x10^{-3}\ kg$$

The mass of contaminant entering the water body from the storm runoff in the adsorbed form, P_{xt} (Equation 9-21), and the dissolved form, P_{qt} (Equation 9-22), is obtained as follows:

$$P_{xt} = \frac{7.56\text{x}10^{5}\ \text{kg}}{1500\ \text{mg/m}^{3} \cdot 1.00\ \text{cm}} \cdot 4.71\text{x}10^{-3}\ \text{kg/km}^{2} \cdot \frac{10^{5}\ \text{cm}}{\text{km}} \cdot \frac{\text{km}^{2}}{10^{9}\ \text{m}^{3}}$$

$$P_{xt} = 2.38\text{x}10^{-4}\ \text{kg}$$

$$P_{qt} = \frac{1.19\ \text{cm}}{(5.0\ \text{cm} + 0.00\ \text{cm})} \cdot 9.13\text{x}10^{-4}\ \text{kg/km}^{2} \cdot 15\ \text{km}^{2}$$

$$P_{qt} = 3.25\text{x}10^{-3}\ \text{kg}$$

The total amount of the chemical in the dissolved state, $CHEM_d$ (Equation 9-13), and the adsorbed form, $CHEM_a$ (Equation 9-14), can be determined by first calculating the total amount of the contaminant entering the water body from all sources, TOTCHEM (Equation 9-19), and multiplying by the distribution factors:

$$\text{TOTCHEM} = 2.38\text{x}10^{-4}\ \text{kg} + 3.25\text{x}10^{-3}\ \text{kg} + 1.03\text{x}10^{-3}\ \text{kg}$$

$$\text{TOTCHEM} = 4.52\text{x}10^{-3}\ \text{kg}$$

$$CHEM_d = 4.52\text{x}10^{-3}\ \text{kg} \cdot 0.186 \cdot \frac{10^{6}\ \text{mg}}{\text{kg}}$$

$$CHEM_d = 839\ \text{mg}$$

$$CHEM_a = 4.52\text{x}10^{-3}\ \text{kg} \cdot 0.565 \cdot \frac{10^{6}\ \text{mg}}{\text{kg}}$$

$$CHEM_a = 3680\ \text{mg}$$

The final concentrations for the short-term Tier 2 and Tier 3 scenario can now be determined. $CHEM_w$, the dissolved concentration of the contaminant in the water body in question is obtained from the dissolved mass and the appropriate dilution volume (Equation 9-10):

$$CHEM_w = \frac{839\ mg}{8.63x10^4\ m^3} \cdot \frac{m^3}{10^3\ L}$$

$$CHEM_w = 9.72x10^{-6}\ mg/L$$

The mass of adsorbed contaminant per mass of sediment in the water body, $CHEM_{s1}$ (Equation 9-11):

$$CHEM_{s1} = \frac{3680\ mg}{7.56x10^5\ kg}$$

$$CHEM_{s1} = 4.86x10^{-3}\ mg/kg$$

The mass of adsorbed contaminant per volume of water, $CHEM_{s2}$ (Equation 9-12):

$$CHEM_{s2} = \frac{3680\ mg}{8.63x10^4\ m^3} \cdot \frac{m^3}{10^3\ L}$$

$$CHEM_{s2} = 4.26x10^{-5}\ mg/L$$

9.5.2.3. SCENARIO C -- For Scenario C, the lifetime of the facility (timef) changes to 100 years, and the deposition rate of cadmium (Fa) is 11.9 kg/km^2-yr. The calculations of the cadmium concentration for this scenario follow.

The concentration of cadmium in the receiving water for long-term loading in Tier 1, Ci_{Lg1} (mg/L), can be calculated using Equation 9-1 and converting to the proper units:

$$Ci_{Lg1} = \frac{11.9 \text{ kg/km}^3\text{-yr} \cdot (15 \text{ km}^2 + 1 \text{ km}^2)}{3.15\text{x}10^7 \text{ m}^3\text{/yr}} \cdot \frac{10^6 \text{ mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3 \text{ L}}$$

$$Ci_{Lg1} = 6.04\text{x}10^{-3} \text{ mg/L}$$

The concentration in the receiving water for short-term loading in Tier 1 Ci_{Sh1} (mg/L), is obtained by replacing Vf_x with VS_x in Equation 9-1:

$$Ci_{Sh1} = \frac{11.9 \text{ kg/km}^3\text{-yr} \cdot (15 \text{ km}^2 + 1 \text{ km}^2)}{8.63\text{x}10^4 \text{ m}^3\text{/yr}} \cdot \frac{10^6 \text{ mg}}{\text{kg}} \cdot \frac{\text{m}^3}{10^3 \text{ L}}$$

$$Ci_{Sh1} = 2.21 \text{ mg/L}$$

For Tier 2 and 3 calculations, variables that are needed in subsequent equations must first be obtained. These are S (Equation 9-26), D_R (Equation 9-25), Q (Equation 9-28) and q_p (Equation 9-29):

$$S = 2.54 \cdot [(1000/78)-10] = 7.16$$

$$D_R = \frac{[5.0 \text{ cm} + 0 \text{ cm} - (0.2 \cdot 7.16 \text{ cm})]^2}{5.0 \text{ cm} + 0 \text{ cm} + (0.8 \cdot 7.16 \text{ cm})} = 1.19 \text{ cm}$$

$$Q = 15 \text{ km}^2 \cdot 1.19 \text{ cm} = 17.8 \text{ km}^2\text{-cm}$$

$$q_p = \frac{15 \text{ km}^2 \cdot 1.19 \text{ cm} \cdot 5 \text{ cm}}{6.0 \text{ hr} \cdot [5.0 \text{ cm} - (0.2 \cdot 7.16 \text{ cm})]} \cdot \frac{\text{hr}}{3.6\text{x}10^6 \text{ s}} \cdot \frac{\text{km}}{10^5 \text{ cm}} \cdot \frac{10^9 \text{ m}^3}{\text{km}^3}$$

$$q_p = 11.6 \text{ m}^3/\text{s}$$

The mass of eroded soil, X_s, is obtained from the Modified Universal Soil Loss Equation (Equation 9-27):

$$Xs = 2.04 \cdot 10^6 \cdot (17.8 \cdot 11.55)^{0.56} \cdot (0.210)\ (0.179)\ (0.5)\ (1.0)$$

$$X_s = 7.56\text{x}10^5 \text{ kg}$$

Calculations for the long-term scenario can now be initiated. The rate of sediment loss to the receiving water is computed from the Universal Soil Loss Equation (Equation 9-3):

$$X_e = \frac{400}{\text{yr}} \cdot \frac{0.21 \text{ ton}}{\text{acre}} \cdot (0.179)\ (0.5)\ (1.0) \cdot \frac{907.18 \text{ kg}}{\text{ton}} \cdot \frac{\text{acre}}{4.047\text{x}10^{-3} \text{ km}^2}$$

$$X_e = 1.68\text{x}10^6 \text{ kg/km-yr}$$

The first order loss rate, k_1 (Equation 9-6) can be calculated by adding the loss rates due to infiltration, k_{1I} (Equation 9-7), erosion, k_{1E} (Equation 9-8) and degradation, k_{1D} (Equation 9-9):

$$k_{1I} = \frac{1\ \text{cm/yr}}{0.22\ \text{mL/cm}^3 \cdot 1.0\ \text{cm} \cdot [1 + (1500\ \text{mg/m}^3 \cdot 0.500\ \text{m}^3\text{/kg}) \div 0.22\ \text{mL/cm}^3)]}$$

$$k_{1I} = 1.33\text{x}10^{-3}\ \text{yr}^{-1}$$

$$k_{1E} = \frac{1.68\text{x}10^6\ \text{kg/km-yr}}{1500\ \text{mg/m}^3 \cdot 1.0\ \text{cm}} \cdot \frac{0.500\ \text{m}^3\text{/kg} \cdot 1500\ \text{mg/m}^3}{0.22\ \text{mL/cm}^3 + (0.500\ \text{m}^3\text{/kg} \cdot 1500\ \text{mg/m}^3)}$$

$$k_{1E} = 0.112\ \text{yr}^{-1}$$

$$k_{1D} = \frac{\ln 2}{100} = 6.93\text{x}10^{-3}\ \text{yr}^{-1}$$

$$k_1 = 1.33\text{x}10^{-3}\ \text{yr}^{-1} + 0.112\ \text{yr}^{-1} + 6.93\text{x}10^{-3}\ \text{yr}^{-1} = 0.121\ \text{yr}^{-1}$$

The maximum contaminant mass per area of soil, Mm (Equation 9-4) is based on the assumption that the combustor has been in operation long enough to allow a steady state concentration to be reached:

$$Mm = 11.9\ \text{kg/km}^2\text{-yr} \cdot \frac{1 - \exp(-0.121\ \text{yr}^{-1} \cdot 100\ \text{yr})}{0.121\ \text{yr}^{-1}}$$

$$Mm = 98.6\ \text{kg/km}^2$$

The contaminant also enters the water body by direct deposition (Equation 9-5):

$$Mw = 11.9\ kg/km^2\text{-}yr \cdot \frac{1\ km^2}{3.15x10^7\ m^3/yr}$$

$$Mw = 3.78x10^{-7}\ kg/m^3$$

For long-term loading, the receiving water concentration can now be calculated using Equation 9-2:

$$Ci_{Lg23} = \frac{1.68x10^8\ kg/km\text{-}yr \cdot 15\ km^2 \cdot 98.6\ kg/km^2}{3.15x10^7\ m^3/yr \cdot 1500\ kg/m^3 \cdot 1.00\ cm} \cdot \frac{10^5\ cm}{km} \cdot \frac{km^3}{10^6\ m^3}$$

$$+ 3.78x10^{-7}\ kg/m^3 \cdot \frac{10^6\ mg}{kg} \cdot \frac{m^3}{10^3\ L}$$

$$Ci_{Lg23} = 5.66x10^{-3}\ mg/L$$

For short-term loading, the Tier 2 or 3 model calculates the receiving water concentration after a storm event. It takes into account any repartitioning of the contaminant upon dilution. This is initiated by calculating the mean sediment concentration in the water body, C_s, Equation 9-17:

$$C_s = \frac{7.56x10^5\ kg}{8.63x10^4\ m^3/yr} = 8.77\ kg/m^3$$

The fraction that would be present in the dissolved, α_d (Equation 9-15), and adsorbed, α_a (Equation 9-16), state can now be determined:

$$\alpha_d = \frac{1}{1 + (0.500\ m^3/kg \cdot 8.77\ kg/m^3)} = 0.186$$

$$\alpha_a = \frac{0.500\ m^3/kg \cdot 8.77\ kg/m^3}{1 + (0.500\ m^3/kg \cdot 8.77\ kg/m^3)} = 0.814$$

After a storm event, the amount of contaminant entering the water body will be partitioned between the adsorbed, Aa (Equation 9-23), and dissolved, Da (Equation 9-24), state:

$$Aa = [1 \div (1 + \frac{0.220\ mL/cm^3}{0.500\ m^3/kg \cdot 1500\ kg/m^3})] \cdot 7.21x10^{-2}\ kg/km^2$$

$$Aa = 1.50x10^{-1}\ kg/km^2$$

$$Da = [1 \div (1 + \frac{0.500\ m^3/kg \cdot 1500\ kg/m^3}{0.220\ mL/cm^3})] \cdot 7.21x10^{-2}\ kg/km^2$$

$$Da = 2.89x10^{-2}\ kg/km^2$$

The amount of chemical deposited directly onto the water body is a function of the deposition rate, area of the water body, and time (Equation 9-20):

$$P_{ft} = 11.9\ kg/km^2\text{-}yr \cdot 2.74x10^{-3}\ yr \cdot 1.0\ km^2$$

$$P_{ft} = 3.26x10^{-2}\ kg$$

The mass of contaminant entering the water body from the storm runoff in the adsorbed form, P_{xt} (Equation 9-21), and the dissolved form, P_{qt} (Equation 9-22), is obtained as follows:

$$P_{xt} = \frac{7.56\text{x}10^{5}\ \text{kg}}{1500\ \text{mg/m}^3 \cdot 1.00\ \text{cm}} \cdot 1.50\text{x}10^{-1}\ \text{kg/km}^2 \cdot \frac{10^5\ \text{cm}}{\text{km}} \cdot \frac{\text{km}^2}{10^9\ \text{m}^3}$$

$$P_{xt} = 7.52\text{x}10^{-3}\ \text{kg}$$

$$P_{qt} = \frac{1.19\ \text{cm}}{(5.0\ \text{cm} + 0.00\ \text{cm})} \cdot 9.13\text{x}10^{-4}\ \text{kg/km}^2 \cdot 15\ \text{km}^2$$

$$P_{qt} = 1.03\text{x}10^{-1}\ \text{kg}$$

The total amount of the chemical in the dissolved state, $CHEM_d$ (Equation 9-13), and the adsorbed form, $CHEM_a$ (Equation 9-14), can be determined by first calculating the total amount of the contaminant entering the water body from all sources, TOTCHEM (Equation 9-19), and multiplying by the distribution factors:

$$\text{TOTCHEM} = 7.52\text{x}10^{-3}\ \text{kg} + 1.03\text{x}10^{-1}\ \text{kg} + 3.26\text{x}10^{-2}\ \text{kg}$$

$$\text{TOTCHEM} = 1.43\text{x}10^{-1}\ \text{kg}$$

$$CHEM_d = 1.43\text{x}10^{-1}\ \text{kg} \cdot 0.186 \cdot \frac{10^6\ \text{mg}}{\text{kg}}$$

$$CHEM_d = 2.66\text{x}10^{4}\ \text{mg}$$

$$CHEM_a = 1.43\text{x}10^{-1}\ \text{kg} \cdot 0.565 \cdot \frac{10^6\ \text{mg}}{\text{kg}}$$

$$CHEM_a = 1.16\text{x}10^{5}\ \text{mg}$$

The final concentrations for the short-term Tier 2 or 3 scenario can now be determined. $CHEM_w$, the dissolved concentration of the contaminant in the water body in question is obtained from the dissolved mass and the appropriate dilution volume (Equation 9-10):

$$CHEM_w = \frac{2.66x104 \ mg}{8.63x10^4 \ m^3} \cdot \frac{m^3}{10^3 \ L}$$

$$CHEM_w = 3.08x10^{-4} \ mg/L$$

The mass of adsorbed contaminant per mass of sediment in the water body, $CHEM_{s1}$ (Equation 9-11):

$$CHEM_{s1} = \frac{1.16x10^5 \ mg}{7.56x10^5 \ kg}$$

$$CHEM_{s1} = 1.54x10^{-1} \ mg/kg$$

The mass of adsorbed contaminant per volume of water, $CHEM_{s2}$ (Equation 9-12):

$$CHEM_{s2} = \frac{1.16x10^5}{8.63x10^4 \ m^3} \cdot \frac{mg/m^3}{10^3 \ L}$$

$$CHEM_{s2} = 1.35x10^{-3} \ mg/L$$

The final results for the concentration of the contaminant in the water body for the A, B and C scenarios are summarized in Table 9-9.

TABLE 9-9

Cadmium Concentration in Receiving Water for the Three Different Scenarios

	A	B	C
Fallout (kg/m^3-yr)	8.94×10^{-3}	3.76×10^{-1}	11.9
Facility lifetime(yr)	30	60	100
Ci_{Lg1} (mg/L)	4.54×10^{-6}	1.91×10^{-4}	6.04×10^{-3}
Ci_{Sh1} (mg/L)	1.66×10^{-3}	6.97×10^{-2}	2.21
Ci_{Lg23} (mg/L)	4.14×10^{-6}	1.79×10^{-4}	5.66×10^{-3}
$CHEM_w$ (mg/L)	2.26×10^{-7}	9.72×10^{-6}	3.08×10^{-4}
$CHEM_{s1}$ (mg/kg)	1.13×10^{-4}	4.86×10^{-3}	1.54×10^{-1}
$CHEM_{s2}$ (mg/L)	9.93×10^{-7}	4.26×10^{-5}	1.35×10^{-3}

10. DETERMINING EXPOSURE FROM WATER INGESTION

10.1. INTRODUCTION

Contaminants from combustor emissions that have been transported into water can be directly ingested when humans drink water. Contaminant intake from water is proportional to the water concentration of the contaminant and the water ingestion rate.

10.2. DAILY INTAKE FROM WATER (DI)

This methodology assesses only risk associated with the increase in exposure due to combustor emissions; the background concentration of contaminants is not included in the quantitation of DI. DI is calculated as follows:

$$DI = Wc \cdot Cw \qquad \text{(Equation 10-1)}$$

where:

DI = total daily intake from water (mg/kg/day)
Wc = water concentration (mg pollutant/L)
Cw = consumption rate of water (L/kg BW/day)

To determine if the contaminant adversely affects human health, the DI is compared with the reference dose for chronic oral exposure to systemic toxicants, such as for cadmium, or used to determine excess risk for carcinogens, such as for benzo(a)pyrene. This will be discussed in Chapter 15.

10.2.1 Water Concentration (Wc). The methods for calculating contaminant concentration in both surface water and collected precipitation are presented in Chapter 9. The long-term Tier 2/3 water concentrations of benzo(a)pyrene (1.55×10^{-7} mg/L) and cadmium (1.79×10^{-4} mg/L) are used for the example calculations in Section 10.3. These values represent the total concentration of benzo(a)pyrene or cadmium in water; concentration is not partitioned into dissolved and adsorbed contaminant. Thus, these values may overestimate exposure because contaminants adsorbed to particles suspended in water would be filtered out during treatment and would not contribute to exposure.

10.2.2. Water Consumption Rate (Cw). The U.S. EPA has traditionally used a value of 2 L/day to represent the average daily water consumption of adults and 1 L/day to represent the average daily water consumption of children with body weights of ≤10 kg. However in a recent document, U.S. EPA (1989a) presents data that suggest these values may overestimate the actual water consumption rates. These data, generated in several consumption surveys, show that a value of 1.4 L/day may be a more accurate average of water consumption for adults. A value of 2 L/day most likely represents a 90th percentile value. U.S. EPA suggests that a value ranging from 0.5 to 0.8 L/day may be a more accurate ingestion rate for children age 5 to 14. In addition, for children with body weights of ≤10 kg, the ingestion rate may be closer to 0.2 L/day.

The daily water ingestion rates used in the example calculations are expressed in the units L/kg BW/day because it can be used for estimating ingestion rates for children over a broad range of ages. The values selected for Scenarios A, B, and C represent approximately the 50th, 70th and 90th percentiles, respectively.

For children, the approximate 90th percentile of 0.1 L/kg/day is used in Scenario C. This is based on the U.S. EPA assumption of 1 L/day ingestion for children with body weights of 10 kg. The approximate 50th percentile of 0.032 L/kg/day is used in Scenario A. This value, presented in U.S. EPA (1989b), is based on an average water ingestion for children 2 years old weighing 13.5 kg (Nelson et al., 1969). For Scenario B, a value of 0.066 L/kg/day was chosen as a reasonable estimate of the 70th percentile.

For adults, the approximate 90th percentile of 0.029 L/kg/day is used for Scenario C. This value is based on a water ingestion of 2 L/day divided by a body weight of 70 kg. The approximate 50th percentile of 0.020 L/kg/day is used for Scenario A. This value is based on a water ingestion of 1.4 L/day divided by a body weight of 70 kg. For Scenario B, a value of 0.024 L/kg/day was chosen as a reasonable estimate of the 70th percentile.

10.3. EXAMPLE CALCULATIONS

This section illustrates a water ingestion exposure to benzo(a)pyrene and cadmium represented by Scenario B. As discussed in Chapter 2, Scenario B is defined as a child who grows up in an area of moderate deposition and remains in the area for 30 years.

A lake in the area is the only source of drinking water. The different exposures using child and adult consumption rates are shown below. Adjustments to exposure are presented in Chapter 15. Exposure due to ingestion of contaminated drinking water can be calculated according to Equation 10-1:

B(a)P

Child's DI $= 1.55\times10^{-7}$ mg/L · 0.066 L/kg/day
$= 1.02\times10^{-8}$ mg/kg/day

Adult's DI $= 1.55\times10^{-7}$ mg/L · 0.024 L/kg/day
$= 3.72\times10^{-9}$ mg/kg/day

Cadmium

Child's DI $= 1.79\times10^{-4}$ mg/L · 0.066 L/kg/day
$= 1.18\times10^{-5}$ mg/kg/day

Adult's DI $= 1.79\times10^{-4}$ mg/L · 0.024 L/kg/day
$= 4.30\times10^{-6}$ mg/kg/day

Chapter 15 shows how these values are used to determine if exposure to combustor emissions poses a human health risk.

11. DETERMINING EXPOSURE FROM FISH INTAKE

11.1. INTRODUCTION

Once contaminants from combustor emissions are transported to surface water bodies, they may be incorporated into the aquatic food chain. Humans may then be exposed to contaminants when they eat fish raised in contaminated surface water bodies. The concentration of contaminants in fish is determined by multiplying the water concentration by a bioconcentration factor, which is the ratio of contaminant concentration in fish to contaminant concentration in water. However, for contaminants with low water solubility that are adsorbed to suspended particles (for example, dioxin and PCBs), use of the fish-water bioconcentration factor may underestimate contaminant concentration in fish. Use of fish-sediment bioconcentration factors have been suggested (Goeden and Smith, 1989; U.S. EPA, 1988a) to overcome this problem. But because fish-sediment bioconcentration factors are generally unavailable in the literature, this methodology will continue to use fish-water bioconcentration factors to estimate contaminant levels in fish.

11.2. CALCULATING DAILY INTAKE FROM FISH

Human daily intake of contaminants is proportional to the water concentration, the bioconcentration factor and the fish ingestion rate. DI is calculated as follows:

$$DI = Wc \cdot BCF \cdot Cf \qquad \text{(Equation 11-1)}$$

where:

DI = total daily intake from fish (mg/day)
Wc = water concentration (mg/L)
BCF = bioconcentration factor (L/g)
Cf = fish ingestion rate (g/kg/day)

11.2.1. Water Concentration (Wc). Methods for determining water concentration are presented in Chapter 9. The long-term Tier 2/3 concentrations of benzo(a)pyrene (1.55×10^{-7} mg/L) and cadmium (1.79×10^{-4} mg/L) for Scenario B are used in the example calculations in Section 11.3. If fish-sediment bioconcentration factors can be located for use in a site-specific assessment, a contaminant concentration in sediment, as calculated in Chapter 9 ($CHEM_{s1}$), can be used in place of a water concentration to estimate contaminant levels in fish.

11.2.2. Bioconcentration Factor (BCF). Bioconcentration is the ability of living organisms to accumulate substances to higher than ambient level concentrations. The degree to which a chemical accumulates above ambient concentrations in an aquatic organism is indicated by the magnitude of the BCF. Specifically, BCF is defined as the ratio of the contaminant concentration in all or part of an aquatic organism (μg/g fresh weight) to the contaminant concentration in the water body where the organism has been exposed (μg/L). The BCF, therefore, has units of (μg/g)/(μg/L) or L/g. Defined in this way, the bioconcentration factor considers only the absorption of a pollutant by an organism from the ambient water and not consumption of contaminated food or water sources.

Bioconcentration factors are specific for the compound and the species absorbing the compound. The compounds with the greatest tendency to bioaccumulate are those that are lipophilic and resistant to biological degradation. Initial transport into the organism occurs by rapid surface adsorption or partitioning to the lipoprotein layer of cell membranes. Once in the bloodstream, subsequent accumulation of the chemical into particular compartments of the organism is dependent upon the metabolic capabilities of the organism and the lipid content of the individual organs. With continuous exposure to a compound, the rate of excretion eventually equals the rate of uptake.

Bioconcentration factors can be estimated through laboratory experiments, field studies, correlations with physicochemical factors such as octanol/water partition coefficients, and models based upon pollutant biokinetics coupled to fish energetics. In the development of the ambient water quality criteria (AWQC), the U.S. EPA used mostly laboratory data in the calculation of BCFs. Field data are often less reliable than laboratory data because it cannot usually be shown that constituent concentrations in field situations have been held constant for a long period of time or over the range of territory inhabited by the organism. BCFs calculated from field data also may be greater than those calculated from laboratory data, which is apparently due to ingestion of the compound through prey, sediments and water, in addition to absorption from water.

When laboratory and field data are not available, BCFs can be estimated by several methods. Correlations between BCFs and octanol/water partition coefficients, water solubility and soil adsorption coefficients have been documented. Veith et al. (1979) developed the following equation using the correlation between the BCF and the n-octanol/water partition coefficient (K_{ow}) to estimate BCFs to within 60% before laboratory testing:

$$\log BCF = 0.85 \log K_{ow} - 0.70 \quad \text{(Equation 11-2)}$$

The equation was developed using data from whole-body analyses of ≈7.6 percent lipids (U.S. EPA, 1980e). The U.S. EPA adopted Equation 11-2 for determining BCFs in the exposure sections of the health effects chapter of the AWQC documents when an appropriate BCF is not available. In a later study, Veith et al. (1980) used the results of their own laboratory experiments and data from other laboratories for a variety of fish species and 84 different organic chemicals to obtain the following modification of their original equation:

$$\log BCF = 0.76 \log K_{ow} - 0.23 \quad \text{(Equation 11-3)}$$

Equations similar to the ones developed by Veith et al. (1979, 1980) have been developed for more specific chemical classes and particular aquatic species (Veith et al., 1979; Neeley et al., 1974). Other investigators (Norstrom et al., 1976) have developed more elaborate models using pollutant biokinetics and fish energetics in addition to using octanol/water partition coefficients to predict BCFs. Values of fish-water BCF and log

K_{ow} for many compounds of environmental concern are available in U.S. EPA (1986d). A BCF of 3.6×10^{-2} L/g for benzo(a)pyrene is used in the example calculations in Section 11.3. This value is a weighted average of values for fish (3.0×10^{-2} L/g) and molluscs (7.9×10^{-2} L/g) (U.S. EPA, 1980b), weighted according to relative consumption of fish (87.8%) and molluscs (12.2%). Thus, the BCF for benzo(a)pyrene would be higher for individuals whose exposure is through a higher percentage of molluscs or other invertebrates. A weighted average BCF for cadmium was calculated to be 6.4×10^{-2} L/g (U.S. EPA, 1980c).

Since bioconcentration for lipophilic compounds depends on lipid content of the fish, it is important to adjust measured or estimated BCF values for these compounds to reflect the lipid content of seafood in the U.S. diet. The U.S. EPA determined in 1980 that the average lipid content of freshwater and estuarine species, weighted by average daily consumption, was 3.0% (U.S. EPA, 1980e). Since fresh and estuarine waters would be those affected by runoff from areas of MWC deposition, a lipid content of 3% should be assumed. The adjustment is made as follows:

$$BCF_a = BCF_u \cdot \frac{LC_d}{LC_e} \qquad \text{(Equation 11-4)}$$

where:

BCF_a = adjusted BCF (L/kg)
BCF_u = unadjusted BCF (L/kg)
LC_d = lipid content of dietary seafood (kg/kg)
LC_e = lipid content of experimental organism (kg/kg)

11.2.3. Fish Ingestion Rate (Cf). Several recent publications have provided estimates of daily fish intake averaged over the entire population, including fish-eaters and non fish-eaters. The most recent fish consumption document from the U.S. Department of Commerce (1985) reports total per capita fish and shellfish consumption ranging from 12.8 pounds/year in 1980 to 13.6 pounds/year in 1984. The latter value is equivalent to a daily intake of 16.9 g of fish (all kinds).

U.S. EPA (1980d) presents data showing means and 95th percentiles of monthly fish consumption for fish consumers in the United States. For adults, the mean consumption rate was 14.3 g/day and the 95th percentile consumption was 41.7 g/day. For children age 0-9 years (16.46 kg BW), the mean consumption rate was 6.2 g/day and the 95th percentile consumption rate was 16.5 g/day. This survey does not distinguish between commercially and recreationally caught fish.

Runoff containing contaminants from combustor emissions could affect freshwater and estuarine species, but would not affect the marine species that constitute the greater portion of seafood in the U.S. diet. To estimate average daily consumption of freshwater fish, the U.S. EPA (1980d) eliminated all fish species from the survey that were not taken from fresh or estuarine waters (Stephan, 1980). This reduced per capita consumption from 14.3 to 6.5 g/day. Therefore, it seems reasonable to assume that in most instances freshwater and estuarine species will constitute 50% of total consumption.

There was a great disparity in fish consumption rates from the national average, depending on region, age, race and religion. The New England and Middle Atlantic regions had the highest fish consumption rates with means of 16.3 and 16.2 g/day, respectively and 95th percentile values of 46.5 and 47.8 g/day, respectively. Based on age, the highest consumption rates were reported for people 60-69 years, who had a mean consumption of 21.7 g/day and 95th percentile consumption of 55.4 g/day. SRI reported fish intake by the Black and Jewish populations to be double the average value (U.S. EPA, 1980d). The highest consumption rate, based on race, was for Orientals, who had a mean consumption of 21.0 g/day and 95th percentile consumption of 67.3 g/day.

The data discussed to this point have included both commercially caught fish and data from non fish-eaters. Commercially caught fish are usually distributed widely so that prediction of consumption from a particular source is difficult. Including data from non fish-eaters underestimates the actual consumption rates for groups eating significant amounts of fish, such as recreational fishermen. Thus, the data discussed to this point may not be accurate for assessing exposure of recreational fishermen from consumption of contaminated fish from one location.

U.S. EPA (1989a) presents data from two surveys that specifically studied consumption rates of recreational fishermen. Although both studies were conducted on the west coast, U.S. EPA (1989a) concludes that they are representative of "consumption rates of recreational fishermen where there is a large water body present and widespread contamination is evident". The 50th, 70th, and 90th percentile consumption rates reported for recreational fishermen are 30 g/day, 80 g/day, and 140 g/day respectively. U.S. EPA (1989a) notes that these values are not recommended "for small water bodies or for areas of localized contamination in large water bodies". For assessments in these areas, U.S. EPA (1989a) recommends that assessors interview local recreational fishermen to derive site-specific data.

The daily fish ingestion rates used in the example calculations are expressed in the units g/kg BW/day because this approach can be used for estimating children's ingestion rates over a broad range of ages. For calculating children's exposure due to fish ingestion, this document uses the average or 95th percentile rates for children age 0-9 years, based on total U.S. population (U.S. EPA, 1980d). The average and 95th percentile rates were adjusted by 50% to account for the fraction of total consumption that can be attributed to fresh water or estuarine sources. The adjusted rates were divided by the average body weight for children age 0-9 years, 16.46 kg (Nelson et al., 1969). The resulting value of 0.188 g/kg BW/day could be used for Scenario A and 0.501 g/kg BW/day could be used for Scenario C. An intermediate value of 0.345 g/kg BW/day was chosen as a reasonable estimate of the 70th percentile and is used in the examples of Scenario B.

For calculating adults' exposure due to fish ingestion, this document uses the U.S. population average consumption rates (14.3 g/day) for Scenario A, and the U.S. population 95th percentile consumption rates (41.7 g/day) for Scenario B. The 75th percentile consumption rate of recreational fishermen (80 g/day) is used for

Scenario C. All values are adjusted by 50% to account for the fraction of total consumption that can be attributed to fresh water or estuarine sources, then divided by 70 kg BW. The resulting values for Scenario A, B, and C are 0.102, 0.298, and 0.571 g/kg BW/day, respectively.

11.3. EXAMPLE CALCULATIONS

This section illustrates exposure to benzo(a)pyrene and cadmium via fish ingestion for Scenario B. As discussed in Chapter 2, Scenario B is represented by a child who grows up in an area of moderate deposition of emissions and remains in the area for 30 years. A lake in the area is used for recreational fishing. The different exposures using child and adult consumption rates are shown below. Adjustments to exposure are presented in Chapter 15. The values of input variables chosen for Scenario B are presented in Table 11-1.

11.3.1. Benzo(a)Pyrene. Exposure due to fish ingestion for a child can be calculated according to Equation 11-1:

$$\begin{aligned} DI &= 1.55\text{x}10^{-7}\ \text{mg/L} \cdot 3.6\text{x}10^{-2}\ \text{L/g} \cdot 0.345\ \text{g/kg/day} \\ &= 1.93\text{x}10^{-9}\ \text{mg/kg/day} \end{aligned}$$

TABLE 11-1

Values for Input Variables Used in Scenario B Example

Input Variable	B(a)P	Cadmium
Wc (mg/L)	1.55×10^{-7}	1.79×10^{-4}
BCF (L/g)	3.6×10^{-2}	6.4×10^{-2}
Cf (g/kg/day)	0.345 - child 0.298 - adult	0.345 - child 0.298 - adult

Exposure due to fish ingestion for an adult can also be calculated according to Equation 11-1:

$$\begin{aligned} DI &= 1.55\text{x}10^{-7}\ \text{mg/L} \cdot 3.6\text{x}10^{-2}\ \text{L/g} \cdot 0.298\ \text{g/kg/day} \\ &= 1.66\text{x}10^{-9}\ \text{mg/kg/day} \end{aligned}$$

These values are used in Chapter 15 to determine excess risk for exposure to benzo(a)pyrene in combustor emissions.

11.3.2. Cadmium. Exposure due to fish ingestion for a child can be calculated according to Equation 11-1:

$$\begin{aligned} DI &= 1.79\text{x}10^{-4}\ \text{mg/L} \cdot 6.4\text{x}10^{-2}\ \text{L/g} \cdot 0.345\ \text{g/kg/day} \\ &= 3.95\text{x}10^{-6}\ \text{mg/kg/day} \end{aligned}$$

Exposure due to fish ingestion for an adult can also be calculated according to Equation 11-1:

$$\begin{aligned} DI &= 1.79\text{x}10^{-4}\ \text{mg/L} \cdot 6.4\text{x}10^{-2}\ \text{L/g} \cdot 0.298\ \text{g/kg/day} \\ &= 3.41\text{x}10^{-6}\ \text{mg/kg/day} \end{aligned}$$

These values are compared with the RfD_o according to the approach described in Chapter 15.

12. DETERMINING EXPOSURE FROM DERMAL ABSORPTION FROM WATER

12.1. INTRODUCTION

Once contaminants in combustor emissions reach water sources, as described in Chapter 9, humans may be exposed by absorbing them through the skin while swimming or bathing. Dermal absorption could be a significant source of exposure from contaminated water compared with ingestion. Brown et al. (1984) reported that dermal absorption of volatile organic contaminants in drinking water accounts for an average of 64% of the dose incurred. Many of the factors that influence dermal absorption from soil also influence dermal absorption from water and the uncertainties associated with the soil pathway (see Chapter 7) can be assumed to apply to the water pathway as well.

12.2. DAILY DERMAL INTAKE FROM WATER

The daily dermal intake from water represents the increase above background in daily human dermal intake due to combustor emissions. As with absorption from soil, exposure due to dermal absorption from water is likely to be influenced by the contaminant concentration in water and the extent of contact. Chapter 9 discusses the procedures for determination of pollutant water concentration. The extent of contact is affected by contact time, frequency and surface area. In addition, the fraction of the contaminant absorbed from water will influence the overall dermal exposure. These factors are presented below.

12.2.1. Contact Time and Frequency. As with dermal exposure to soil, contact time is defined as the amount of time each day skin is in contact with contaminated water; frequency is defined as the number of days per year that skin is in contact with contaminated water. Skin comes into contact with contaminated water when individuals swim in contaminated surface water sources or bathe in contaminated water that originates from either a surface water source or collected precipitation.

U.S. EPA (1988b) reports data quantifying the contact time and frequency of swimming, based on information from the Bureau of Outdoor Recreation. The average contact time for swimming is 2.6 hours/day, and the average frequency is 7 days/yr. However, these averages are likely to vary significantly depending on local climate and location. Thus, risk assessors should consult the local recreation department for site-specific data when conducting an assessment.

U.S. EPA (1989a) reports data quantifying the contact time and frequency of bathing. The 50th percentile bathing time is approximately 0.12 hours/day (7 minutes); the 75th percentile bathing time is approximately 0.15 hours/day (9 minutes); the 90th percentile bathing time is approximately 0.2 hours/day (12 minutes). Ninety percent of Americans report bathing once a day, and 5% report bathing more than once a day.

12.2.2. Surface Area. Dermal intake of a contaminant is proportional to the exposed surface area. For exposure via swimming and bathing, the exposed surface area is assumed to be the total body surface area. U.S. EPA (1989a) presents total body surface area data for adult males, adult females and children.

A quantitative approach to estimating dermal exposure to contaminants in water is precluded by the paucity of data available concerning dermal absorption from water.

13. HAZARD IDENTIFICATION

13.1. INTRODUCTION

The purpose of hazard identification, the first step in the risk assessment process, is to determine the type of toxicity that is produced by a chemical agent (e.g., mutagenicity, developmental toxicity, carcinogenicity). Hazard identification is the qualitative risk assessment in which all available experimental animal and human data are reviewed to determine if the agent is likely to cause a particular type of toxicity.

The hazard identification step in this document distinguishes two types of toxic responses, i.e., whether or not a chemical agent has the potential to increase the incidence of cancer in exposed humans, because the U.S. EPA's approaches to assessing risks associated with carcinogens and noncarcinogens are different. Pollutants emitted from combustors may include carcinogens and noncarcinogens.* The purpose of this chapter is to describe the general process of hazard identification and to show how the process can be used in evaluating the hazard of toxic pollutants in combustor emissions.

In the analysis of data regarding the potential human carcinogenicity of chemical agents, the U.S. EPA uses the approach described in its Guidelines for Carcinogen Risk Assessment (U.S.

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*The decision as to whether to treat a chemical as a threshold- or nonthreshold-acting (carcinogenic) agent depends on the weight of evidence that it may be carcinogenic to humans (refer to section 13.3).

EPA, 1986f). This follows the general format of the National Academy of Sciences (NRC, 1983) in its description of the risk assessment process.

For carcinogenic chemicals, the Agency considers the risk of cancer to be linearly related to dose (except at high dose levels) (U.S. EPA, 1986f). Therefore, while risk diminishes with dose, it does not become zero until dose becomes zero. The decision whether to treat a chemical as a threshold- (systemic toxicant) or nonthreshold-acting (carcinogenic) agent depends on the weight of evidence that it may be carcinogenic to humans. The agent's potential for human carcinogenicity is inferred from the available information relevant to the potential carcinogenicity of the chemical and from judgments as to the quality of the available studies. Chemicals that are not considered to be carcinogenic are assumed to produce systemic effects with a dose threshold.

13.2. ELEMENTS OF HAZARD IDENTIFICATION

All relevant data providing both principal and supporting evidence of the chemicals' toxic effects (e.g., carcinogenic, systemic toxicity) should be included in the hazard identification process. The studies providing principal evidence (particularly for the purposes of this methodology) are those that contribute most significantly to a qualitative assessment of whether or not a particular chemical is potentially carcinogenic in humans. Principal evidence is obtained from: (1) epidemiologic studies or human studies that elaborate the association between cancer incidence and exposure to the agent, and (2) long-term animal studies conducted under controlled laboratory conditions. Supporting evidence is derived from short-term tests for genotoxicity, toxicologic effects other than cancer, metabolic and pharmacokinetic properties, structure-activity relationships, and physical/chemical properties of the agent (U.S. EPA, 1986f).

In the evaluation of noncancer health effects, the U.S. EPA assumes a threshold exists. Hazard identification for noncancer effects involves the consideration of the toxic endpoints from all available studies with primary attention given to identification of a "critical effect". Principal studies, both human studies and animal studies, contribute to the qualitative assessment of whether an agent is potentially a systemic toxicant in humans (Barnes and Dourson, 1988). These studies are evaluated for quality and adequacy using many of the same criteria as for carcinogenicity studies.

13.2.1. Human Studies/ Epidemiologic Studies. Data from human studies should be used to qualitatively establish the presence of an adverse effect in exposed human populations when such information is available. The strength of the human evidence for both carcinogenicity and systemic toxicity depends, among other things, on the type of analysis and on the magnitude of the response. The weight of evidence increases with the number of adequate studies that show comparable results on populations exposed to the same agent under different conditions (U.S. EPA, 1986f; Barnes and Dourson, 1988).

13.2.2. Animal Studies. For most chemicals, there is a lack of appropriate data on effects in humans. In such cases, the principal studies are drawn from experiments conducted on nonhuman mammals (U.S. EPA, 1986f; Barnes and Dourson, 1988). These long-term studies include those for chronic systemic effects, as well as carcinogenicity.

Information from all of these studies is useful in the hazard identification phase of risk assessment. Animal studies are conducted using a variety of exposure durations (e.g., acute, subchronic, and chronic), schedules (e.g., single, intermittent, or continuous dosing), and routes (e.g., oral, dermal, inhalation), and may be used to further support an identified toxic/carcinogenic response. For example, overt neurologic problems identified in high-dose acute studies tend to reinforce the observation of subtle neurologic changes seen in low-dose chronic studies. Special attention is given to studies involving low-dose, chronic exposures, since such exposures can elicit effects absent in higher dose, shorter exposures, through mechanisms such as accumulation of toxicants in the organisms (Barnes and Dourson, 1988).

13.2.3. Supportive Evidence. These studies provide supportive, rather than definitive, information and can include data from a wide variety of sources. For example, in vitro studies can provide insights into the chemical's potential for biological activity; in certain circumstances, consideration of structurally-related compounds can provide clues to the test chemical's possible toxicity. Similarly, relevant metabolic and pharmacokinetic studies can provide insights into the mechanism of action of a particular compound (U.S. EPA, 1986f; Barnes and Dourson, 1988).

13.2.3.1. SHORT-TERM AND OTHER TOXICITY STUDIES -- Short-term tests for mutagenicity, including tests for point mutations, numeric and structural chromosome aberrations, DNA damage/repair, and in vitro transformation provide supportive evidence of carcinogenicity and may give information on potential carcinogenic mechanisms. Short-term in vivo and in vitro tests that can give an indication of initiation and promotion activity may also provide supportive evidence for carcinogenicity. Lack of positive results in short-term tests for genetic toxicity does not provide a basis for discounting positive results in long-term animal studies (U.S. EPA, 1986f).

Toxicologic effects other than carcinogenicity or mutagenicity, including systemic, teratogenic, and other reproductive effects may also be relevant to the evaluation of carcinogenicity. Prechronic and chronic toxicity evaluations and dose-response and time-to-response analyses of these reactions may yield information on target organ effects, pathophysiologic reactions, and preneoplastic lesions that bear on the evaluation of carcinogenicity (U.S. EPA, 1986f).

13.2.3.2. PHYSICAL-CHEMICAL PROPERTIES -- Chemical specific parameters relevant to carcinogenesis, including physical state and physical-chemical properties, and pertinent information on whether structure-activity relationships support or argue against a prediction of potential carcinogenicity also should be considered. Additionally, relevant metabolic information such as whether the agent is direct-acting or requires conversion to a reactive carcinogen (e.g., electrophilic) species, metabolic pathways for such conversions, macromolecular interactions, and fate (e.g., transport, storage, and excretion), as well as species differences, should be critically evaluated (U.S. EPA, 1986f). By comparing the metabolism of the chemical exhibiting the toxic effect in the animal with the metabolism found in humans, it may be possible to assess the potential for toxicity or to estimate the equitoxic dose in humans (U.S. EPA, 1986f; Barnes and Dourson, 1988).

13.2.3.3. ROUTES AND PATTERNS OF EXPOSURE -- Since the U.S. EPA often approaches the investigation of a chemical with a route of exposure in mind (e.g., an oral exposure for a drinking water contaminant or an inhalation exposure for an air contaminant), consideration is given to potential differences in absorption or metabolism resulting from different routes of exposure, and whenever appropriate data (e.g., comparative metabolism studies) are available, the impact of these differences on the quantitative risk assessment are delineated. In most cases, the toxicologic data base does not include detailed testing on all possible routes of administration. Generally, the U.S. EPA's position is that the potential for toxicity manifested via one route of exposure is relevant to consideration of any other route of exposure, unless convincing evidence exists to the contrary (U.S. EPA, 1986f; Barnes and Dourson, 1988). This may be an important consideration in this methodology, since exposure may occur by several routes.

13.3. WEIGHT OF EVIDENCE

As the culmination of the hazard identification step, a discussion of the weight of evidence summarizes the information gleaned from the principal and supportive studies. Emphasis is given to examining the results from different studies to determine the extent to which a consistent, plausible picture of toxicity emerges. For example, the weight of evidence involves consideration of the quality and adequacy of the human studies; valid and adequate studies strengthen the evidence for carcinogenicity.

Evaluation of individual studies in humans and animals requires the consideration of several factors associated with a study's hypothesis, design, execution, and interpretation. An ideal study addresses a clearly delineated hypothesis, follows a carefully prescribed protocol, and includes sufficient subsequent analysis to support its conclusions convincingly.

Among the criteria for the adequacy of epidemiologic studies are: proper selection and characterization of exposed and control groups; adequate duration and quality of follow-up; proper identification and characterization of confounding factors and bias; consideration of latency affects; valid ascertainment of the causes of morbidity and death; and the ability to detect specific effects (U.S. EPA, 1986f). In evaluating these studies consideration is also given to characterization of the compound(s) under study, similarities and differences between the test individuals in the study groups, the number of study groups and the types of observations and methods of statistical analysis.

Animal studies add to the weight of evidence that the chemical poses a hazard to humans. The weight of evidence increases with: an increase in number of tissue sites affected by the agent; similar results in replicated animal studies by different investigators; similar effects across sex, strain, species, number of experiments and doses and route of exposure; clear evidence of a dose-response relationship and high level of statistical significance of the increased response in treated compared with control groups; a plausible relationship between data on metabolism, postulated mechanism-of-action, and the toxic endpoint; similar toxicity exhibited by structurally-related compounds; and some link between the chemical and evidence of the critical endpoint in humans (U.S. EPA, 1986f; Barnes and Dourson, 1988). Consideration is also given to the type of test species, the spacing and choice of the dose levels tested, the age and sex of the test animals and the route and duration of exposure. For carcinogenicity studies, dose-related shortening of the time-to tumor occurrence or time-to-death with tumor and the occurrence of a dose-related increase in the proportion of tumors that are malignant also factor into the weight of evidence.

The scheme used to categorize the weight of evidence for the agent in question as a human carcinogen has three major steps. In the first step, the evidence is characterized for human studies and for animal studies. Secondly, the human and animal evidence are combined into a provisional overall classification. In the third stage, the provisional classification is adjusted upwards or downwards, based on analysis of the supporting evidence. The result is that each chemical is placed into one of the following five categories (U.S. EPA, 1986f; U.S. EPA, 1990):

> Group A: Human Carcinogen. Sufficient evidence exists from epidemiology studies to support a causal association between exposure to the chemical and human cancer.
>
> Group B: Probable Human Carcinogen. Sufficient evidence of carcinogenicity in animals with limited (Group B1) or inadequate (Group B2) evidence in humans.
>
> Group C: Possible Human Carcinogen. Limited evidence of carcinogenicity in animals in the absence of human data.
>
> Group D: Not Classifiable as to Human Carcinogenicity. Inadequate human and animal evidence of carcinogenicity or for which no data are available.
>
> Group E: Evidence of Noncarcinogenicity for Humans. No evidence of carcinogenicity in at least two adequate animal tests in different species or in both adequate epidemiologic and animal studies.

Scientific judgment is necessary when deciding which classifications constitute sufficient evidence for basing a quantitative risk assessment on a presumption of carcinogenicity. The quantitative estimates of carcinogenic risk of chemicals in categories A and B at low doses have usually been made with a linear no-threshold dose-response model, whereas those in categories D and E have usually been assessed according to threshold effects. Chemicals classified as C have received varying treatment. The use of the weight of evidence classification, without noting the explanatory material for a specific chemical, may lead to a flawed conclusion since some of the classifications are exposure route dependent.

In evaluating the results from both carcinogenicity and systemic toxicity studies, consideration is given to many other factors, including chemical characterization of the compound(s) under study, the type of test species, similarities and differences between the test individuals in the study groups, the number of study groups, the spacing and choice of dose levels tested, the types of observations and methods of analysis, the nature of pathologic changes, the alteration in metabolic responses, the sex and age of the test animals, and the route and duration of exposure (U.S. EPA, 1986f; Barnes and Dourson, 1988).

13.3.1. EPA Classification of Benzo(a)Pyrene and Cadmium. Benzo(a)pyrene and cadmium have been used throughout this document to illustrate the application of the exposure methodology and will be used as examples for the estimation of human health risk.

Human data specifically linking benzo(a)pyrene to a carcinogenic effect are lacking, even though lung cancer has been shown to be induced in humans by various mixtures of polycyclic aromatic hydrocarbons known to contain benzo(a)pyrene (U.S. EPA, 1990). It is not possible to conclude from this information that benzo(a)pyrene is the only responsible agent. In this case, the human data are considered inadequate for classification of benzo(a)pyrene. However, multiple animal studies in rodent and nonrodent species demonstrating benzo(a)pyrene to be carcinogenic following administration by oral, intratracheal, inhalation and dermal routes lend support to the evidence for carcinogenicity. In addition, benzo(a)pyrene has produced positive results in several in vitro bacterial and mammalian genetic toxicity assays, as well as a number of in vivo tests for DNA damage. It is metabolized to reactive electrophiles capable of binding to DNA. The principal evidence from human and animal studies, along with the supporting evidence resulted in a weight of evidence classification for benzo(a)pyrene of B2, a probable human carcinogen (U.S. EPA, 1990). For illustrative purposes in this document, benzo(a)pyrene will be used as the example of determination of exposure and risk for a carcinogen emitted by combustors.

Cadmium is classified as a probable human carcinogen (B1) (U.S. EPA, 1985c). Insufficient information exists to estimate an oral potency for cadmium (U.S. EPA, 1990). However, an oral RfD is available for cadmium (U.S. EPA, 1990) (see also Chapter 14). Because carcinogens may also produce systemic toxicity and most of the exposures in this document are by oral routes, this document will use cadmium as the example of determination of exposure and risk for a noncarcinogen emitted by combustors.

13.4. SUMMARY AND CONCLUSION

The purpose of the hazard identification step of the risk assessment process is to qualitatively determine the type of toxic response that is produced by a chemical agent. Since the risk assessment process is different for carcinogens and noncarcinogens, for purposes of this document, hazard identification involves determining whether or not the chemical has the potential to increase the incidence of cancer in exposed humans. Principal evidence is obtained from human or epidemiologic studies and long-term, controlled animal studies. Supporting evidence comes from short-term genotoxicity and other toxicity tests, studies on metabolism and other physical-chemical properties, and structure-activity relationships. All of the evidence is evaluated and the chemical is placed in one of the five weight of evidence categories for carcinogenicity according to the primary evidence, with any adjustment to that classification resulting from the supporting evidence.

If the chemical is classified as a carcinogen, then it will be necessary to calculate the excess risk. If a chemical is classified as a noncarcinogen, or threshold-acting agent, then the assessor should determine risk by comparing exposure with a reference dose (RfD) as discussed in Chapter 15.

14. DOSE-RESPONSE ASSESSMENT

14.1. INTRODUCTION

The dose-response assessment is the second step of the risk assessment process (NRC, 1983) and is an attempt to define the relationship between the dose of a chemical and the resulting toxic and/or carcinogenic effects. This assessment results in a quantitative measure of a concentration or dose of a chemical above which humans are at risk for systemic toxicity, or an estimate of carcinogenic potency or risk when humans are exposed for a lifetime. In the case of systemic toxicity, this dose is termed the reference dose, RfD; in the case of carcinogenicity, the estimate of carcinogenic potency is termed the slope factor. Many RfD and cancer slope factor values are currently available on the U.S. EPA Integrated Risk Information System (IRIS) (U.S. EPA, 1990). Chemicals that produce systemic toxicity are assumed to have a threshold dose below which no adverse, noncarcinogenic health effects occur, and chemicals that produce carcinogenic effects are assumed to act without a threshold. Therefore, the respective dose-response assessment processes are different and will be discussed separately. This chapter presents information necessary for understanding the derivation process.

14.2. DOSE-RESPONSE ASSESSMENT FOR SYSTEMIC TOXICITY

Empiric observation generally reveals that as the dosage of a toxicant increases, the toxic response also increases. In general, this increase occurs in severity, intensity and frequency within the population. Such dose-response relationships are well-founded in the theory and practice of toxicology and pharmacology (Barnes and Dourson, 1988).

14.2.1. Selection of the Critical Data. Before an RfD can be calculated for a given chemical, the pertinent information must be selected from the available literature. Because of the difficulties inherent in evaluating a dose-response relationship for a given chemical, scientific judgment must be exercised during the selection process. There must be a decision as to the critical endpoint to measure as the "response". There must be a determination of the correct measure of "dose". Other factors that must be considered include interspecies variation, appropriate units for dose, and the concept of administered dose versus absorbed dose versus target organ dose (Barnes and Dourson, 1988).

The NOAEL (no-observed-adverse-effect level) is defined as the highest experimental dose of a chemical at which there is no statistically or biologically significant increase in frequency or severity of an adverse effect in individuals in an exposed group when compared with individuals in an appropriate control group. Sometimes a chemical may induce more than one adverse effect (endpoint) in the same test animal. In this case, the critical effect selected for the dose-response assessment is the effect resulting from the lowest NOAEL. The NOAEL is the most important information obtained from the study of the dose-response relationship and is the measurement on which the quantitative assessment of the human risk from systemic toxicants is based (Barnes and Dourson, 1988).

Risk assessments based on human data have the advantage of avoiding the problems inherent in interspecies extrapolation. An appropriate human study with a well-defined NOAEL for a critical effect has preference over animal studies as a basis for the RfD determination. For most chemicals, human cohort investigations of noncancer endpoints have not been conducted; however, data from retrospective epidemiologic studies can be used when available. In some cases, a well-designed and well-conducted epidemiologic study that shows no association between known exposures and toxicity can be used to determine an RfD. However, in many instances, use of epidemiologic studies involves extrapolation from relatively high doses (such as those found in occupational settings) to the low doses found in the environmental situations to which the general population is more likely to be exposed (Barnes and Dourson, 1988).

When human data are not available, the most appropriate NOAEL of the critical effect from a well-conducted study on an animal species that is known to resemble the human in response to the particular chemical is chosen. When the most similar species cannot be determined, the most sensitive species (i.e., the species showing a toxic effect at the lowest administered dose) is selected. Use of data from controlled studies of genetically homogeneous animals requires extrapolation from animals to humans and from high experimental doses to comparatively low environmental exposures, as well as consideration of human heterogeneity and possible concurrent human exposures to other chemicals (Barnes and Dourson, 1988).

14.2.2. Reference Dose (RfD). The RfD is a bench mark dose operationally derived from the NOAEL by consistent application of uncertainty factors (UFs) that reflect various types of data sets used to estimate RfDs (Barnes and Dourson, 1988). It is defined as an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. The RfD is calculated as follows:

$$\mathrm{RfD} = \frac{\mathrm{NOAEL}}{\mathrm{UF} \times \mathrm{MF}} \qquad \text{(Equation 14-1)}$$

where:

NOAEL = no-observed-adverse-effect level (mg/kg/day)
UF = uncertainty factor (unitless)
MF = modifying factor (unitless)

The RfD is useful as a reference point from which to judge the potential effects of the chemical at other doses. Usually, doses less than the RfD are not likely to be associated with adverse health risks, and are therefore less likely to be of regulatory concern. As the frequency and/or magnitude of the exposures exceeding the RfD increase, the probability of adverse effects in a human population increases. However, it should not be concluded that all doses below the RfD are "safe" or without risk and that all doses greater than the RfD are "unsafe" or pose a risk to human health (Barnes and Dourson, 1988).

The oral RfD (RfD_o) is expressed in units of milligrams per kilogram of bodyweight per day (mg/kg/day). Methodology for determination of an RfD for chronic inhalation exposure to a chemical (RfD_i) is currently being developed (U.S. EPA, 1989c). Because this document deals only with indirect exposures, i.e., those resulting from deposition of combustor emissions on environmental media, direct inhalation exposure is not considered. Indirect inhalation exposure such as inhalation of resuspended dust, is not likely to be a significant route of exposure (Figure 2-1).

14.2.3. Selection of Uncertainty and Modifying Factors. Uncertainty factors (UFs) are reductions in the dose rate needed to account for several areas of scientific uncertainty inherent in most toxicity data bases. Currently, uncertainty factors of 10 each are applied to extrapolate from animals to humans, to provide protection for unusually sensitive individuals, to expand from subchronic to chronic exposure, to estimate a NOAEL from a lowest-observed-adverse-effect level (LOAEL) and to reflect deficiencies in the data base.

There may be additional uncertainties in the estimation of an RfD such as scientific uncertainties in the key study, chemical-specific issues, and/or deficiencies in the overall data base. In these instances a modifying factor (MF) greater than zero but ≤10 is applied to account for these considerations. The default value for the MF is 1. The maximum total uncertainty factor applied in the derivation of an RfD is 10,000.

14.2.4. Confidence in the RfD. When adequate data from acceptable human studies are available, their use is preferred as the basis for the RfD determination because the resulting value is more certain, and the confidence in the RfD will be higher. In the absence of human data, a complete data base consisting of chronic and developmental toxicity studies in two mammalian species and a 2-generation reproductive study in one mammalian species is preferred. The existence of such a complete data base would indicate that there is little probability that additional data would affect the RfD and, therefore, confidence in the resultant RfD will be high. In some cases, however, general toxicity information may indicate the need for special studies such as neurotoxicity, immunotoxicity, or reproductive toxicity, making the data base incomplete until these studies are performed. The minimum data base needed for an RfD estimation is a single, well-conducted, subchronic animal study; however, for such a data base, the probability that additional data might change the estimate is high so there is low confidence in the RfD.

14.2.5. Calculation of RfD for Cadmium. Throughout this document, sample calculations have been presented using data for cadmium as an example of a systemic toxicant emitted by a MWC. Thus, for explanatory purposes, it is appropriate to present the calculation of an RfD for this chemical.

A U.S. EPA (1985b) study showed that a concentration of 0.2 mg Cd/g wet human renal cortex was the highest renal level not associated with significant proteinuria. A toxicokinetic model exists with which the level of chronic human oral exposure (NOAEL) resulting in 0.2 mg Cd/g wet human renal cortex can be determined; the model assumes that 0.01% of the cadmium body burden is eliminated per day (U.S. EPA, 1985b). Assuming 2.5% absorption of cadmium from food or 5% from water, the model predicts that the NOAELs for chronic cadmium exposure are 0.01 and 0.005 mg Cd/kg/day from food and from water, respectively (i.e., levels that would result in 0.2 mg Cd/g wet weight human renal cortex). A UF of 10 was selected to account for intrahuman variability to the toxicity of cadmium in the absence of specific data on sensitive individuals. The default MF (1) was used. Thus, the U.S. EPA RfD Work Group has established the oral RfD for cadmium to be 1×10^{-3} mg/kg/day in food and 5×10^{-4} mg/kg/day in drinking water. Because the choice of the NOAEL reflects data obtained from many studies on the toxicity of cadmium in both humans and animals and permits calculation of cadmium pharmacokinetic parameters such as absorption, distribution, metabolism and elimination, confidence in the data base is high, resulting in high confidence in both RfD values (U.S. EPA, 1990).

14.3. DOSE-RESPONSE ASSESSMENT FOR CARCINOGENS

The purpose of the dose-response assessment step in carcinogen risk assessment is to define the relationship between the dose of a chemical and the likelihood of a carcinogenic effect assuming the chemical to be a human carcinogen. The goal of the assessment is to estimate the potency of the chemical and to determine the "slope factor" that is used in conjunction with the exposure estimate to yield a numeric estimate of risk. Dose-response assessment usually involves extrapolation from the high doses administered to experimental animals to the much lower exposure levels expected from human contact with the agent in the environment; it also includes considerations of the validity of these extrapolations.

14.3.1. Selection of Data Sets. The dose-response assessment process requires scientific judgment in the choice of appropriate data sets. As with systemic toxicity determinations (previously discussed), human data are preferable to animal data. In the absence of appropriate human data, data from an animal species whose biologic responses are similar to those of humans would be selected. When this information is not available, data from the most sensitive animal species/strain/sex combination are preferred. The route of administration is also a consideration and, whenever there is a choice, the route that most resembles the route of human exposure is selected. When this is not possible, the route difference is noted as a sign of uncertainty. Also, when the incidence of tumors is significantly elevated at more than one anatomical site, estimates of overall risk are made by determining the number of animals with tumors at one or more of these sites. The number of benign tumors is generally combined with the number of malignant tumors unless they are not considered to have potential to progress to the associated malignancies of the same histogenic origin (U.S. EPA, 1990).

14.3.2. Choice of Extrapolation Model. Since human risk due to a carcinogen at low exposure levels cannot be directly determined, a number of mathematical models and procedures have been developed for use in extrapolating from high to low doses (U.S. EPA, 1990). Extrapolation is ordinarily conducted by fitting a mathematical model to the observed data and then extending the model (or a bound on the risks it predicts) from the observed range down toward risks expected at low exposure.

Cancer risk estimates based on adequate human epidemiologic data are preferred over estimates based on animal studies. However, in most cases, the cancer potency estimate must be determined by applying a low-dose extrapolation model, such as the linearized multistage procedure (the U.S. EPA's default procedure), to animal data. Using this procedure, the slope factor is determined by the upper bound of the 95% confidence limit on the slope. (For the linearized multistage procedure, this is referred to as the q_1^*). The units are per(mg/kg)/day.

When animal data are used as a basis for extrapolation, the human dose that is equivalent to the dose in the animal study is calculated assuming that different species are equally sensitive to the effects of a carcinogen if they absorb the same dose per unit of body surface area. This assumption is made only in the absence of specific information about equivalent doses for the chemical in question (U.S. EPA, 1990).

14.3.3. Route-to-Route Extrapolation. If data for one route of exposure are positive, and there are no or insufficient data for other routes of exposure, the chemical is assumed to be carcinogenic by all routes and the resultant slope factor may be adopted for all routes. In order to evaluate human risks for oral exposure when only inhalation exposure has been tested in animals, the slope factor for inhalation exposure can be adopted as the slope factor for oral exposure with application of an adjustment to reflect route-specific differences in extent of absorption. This is accomplished by multiplying the slope factor by the ratio of percent absorption by the oral route to the percent absorption by the inhalation route if the values are available. If this information is not available, assumptions regarding route-specific differences in absorption are considered on a case-by-case basis, and usually the assumption of equal percent absorption by the two routes is made.

14.3.4. Confidence in Quantitation. The scientific data base used to calculate and support the setting of cancer risk levels has an inherent uncertainty that is due to the systematic and random errors in scientific measurement. In most cases, only studies using experimental animals have been performed. Thus, there is uncertainty when the data are extrapolated to humans. When developing cancer risk levels, several other areas of uncertainty exist, such as the incomplete knowledge concerning the health effects of contaminants in the chemical, the impact of the age, sex, and species of the experimental animals used, the nature of the target organ system(s) examined and the effect of dose at the target site in humans or experimental animals. When there is concurrent exposure to more than one chemical, additional uncertainty results from inadequate information concerning possible synergistic, antagonistic, or masking effects.

14.3.5. Calculation of q_1^* for Benzo(a)pyrene. Throughout this document, sample calculations have been presented using data for benzo(a)pyrene as the example of a carcinogen emitted by a MWC. Thus, for explanatory purposes, it is appropriate to present the calculation of a slope factor for this chemical. A carcinogenicity slope factor for human oral exposure to benzo(a)pyrene has been derived from a bioassay in which the chemical was given to strain CFW mice for ≈110 days at dose levels ranging from 1-250 ppm in the diet (Neal and Rigdon, 1967). From the data, the U.S. EPA (1980b) derived a q_1^* of 11.5 per(mg/kg)/day. This value is not available on IRIS, but has been published in U.S. EPA (1980b).

14.4. SUMMARY

Dose-response assessment for a chemical is an attempt to define the relationship between the dose of a chemical and the resulting toxic and/or carcinogenic effects. The result of this assessment is a quantitative measure of a concentration or dose of a chemical above which humans are at risk for systemic toxicity, or an estimate of carcinogenic potency of the chemical when humans are exposed for a lifetime. In the case of systemic toxicity, this dose is termed the reference dose, RfD; in the case of carcinogenicity, this value is termed the slope factor. The amount of confidence in the values determined for any given chemical depends on the quality of the data on which the calculations are based. The next chapter will use the results of the dose-response assessment to determine a quantitative human risk from exposure to chemicals from combustor facilities.

15. RISK CHARACTERIZATION

15.1. INTRODUCTION

Risk assessments ordinarily proceed from source to receptor. The source is first characterized and contaminant movement away from the source is then modeled to estimate the degree of exposure to the receptor, as described in Chapters 3-12. Health effects are then predicted based on the estimated exposure. Risk characterization is the process by which the significance of risk estimates for exposure is evaluated.

The methodology presented here considers only the effects from indirect exposures, that is, exposures following deposition of pollutants to soil or surface waters. Although direct exposure due to inhalation of emitted pollutants is not addressed in this methodology, the risk characterization process would be similar to that for indirect exposures. Human exposure to a contaminant by ingestion or dermal contact is characterized by estimating daily intake (DI) via food, soil, or water as described in Chapters 4-12. For all exposure pathways, the daily intake represents the increased amount of contaminant above background or existing levels to which the individual is exposed due to operation of the combustion facility. Estimates of daily intake have been determined using various exposure scenarios as shown in Chapter 2. Adjustments to exposure calculated for each pathway may be necessary to more accurately estimate lifetime exposure and evaluate the potential for adverse effects on human health. These possible adjustments to exposure are described in Section 15.2.

Risks for adverse human health effects are estimated differently for systemic toxicants, assumed to be threshold-acting chemicals, and carcinogens, assumed to be nonthreshold-acting chemicals. For threshold-acting toxicants, daily intake is compared with the reference dose for chronic exposure (RfD) to determine if the contaminant poses a risk to human health. The RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of the daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious effects during a lifetime.

For carcinogens, the daily intake is used to estimate excess risk. The excess risk (ER) is defined as the incremental lifetime cancer risk above background occurring in a hypothetical population in which all individuals are exposed continuously to a concentration equal to the daily intake of the contaminant. The ER is derived from the daily incremental dose of the contaminant above background and the human cancer potency factor as established by the U.S. EPA. These approaches to estimating risk are presented in Section 15.3. An uncertainty analysis is described in Section 15.4.

15.2. ADJUSTMENTS TO EXPOSURE

Exposure scenarios developed for a site-specific assessment may involve exposure via several routes and pathways. An individual may be exposed to emitted contaminants by several routes during the time he resides in the vicinity of the combustor facility. Daily intake, however, has been calculated independently for each exposure pathway. In order to evaluate exposure by several routes, the duration of exposure and relative effectiveness of each route need to be considered to determine "total" exposure to the individual.

15.2.1. Relative Effectiveness of Exposure (RE). Relative effectiveness is a unitless factor that reflects the relative toxicologic effectiveness of a chemical by different exposure routes. RE is used to standardize effects of exposure by one route to the effects of exposure by another route. Effectiveness is the manifestation of the same inherent toxicity (same toxic endpoint) by the different routes of exposure. The RE of exposure is an experimentally determined value relating the dosages that produced a particular toxicity. Therefore, RE can be used to express the dose of one exposure route in terms of an equivalent dose of another route. The value of RE may, in part, reflect observed or estimated differences in absorption rates between different exposure routes. It can also be related to differing sensitivity of absorption sites to damage and/or differences in toxicokinetics between exposure routes. In addition to route differences, RE can reflect differences in exposure conditions or differences between sources of exposure for the same route, such as ingestion of food or drinking water.

In order to determine the health risk associated with daily intake by ingestion or dermal exposure to contaminants from combustor emissions, the daily intake must be compared with a toxic threshold or the carcinogenic potency of the contaminant by a similar route of exposure. Before applying an RE adjustment to exposure, the route of administration used for estimation of the RfD or carcinogenic potency factor (such as the q_1^*) must be known. Exposures are adjusted by the RE factors with respect to the same route of exposure as the health-based criteria. For example, if an oral RfD (RfD_o) for the chemical of interest is available, RE factors would be applied to exposure determined by this methodology with respect to the same route of exposure as the study from which the RfD was derived (e.g., ingestion of food). Similarly, exposures would be adjusted by an RE relative to inhalation for comparison with the RfD_i. In cases of exposure from soil ingestion, the RE values should take into account the soil matrix if supporting data are available.

For exposure routes for which health-based criteria are not currently available, such as for dermal exposure to toxicants (i.e., RfD for chronic dermal exposure), the DI can be transformed to an equivalent daily intake for the same route for which health-based criteria are available. In the case of dermal exposure, RE refers to the effectiveness of the absorbed dermal dose relative to an unabsorbed ingested dose. This is because daily dermal intake (DDI) is an estimate of absorbed dose, whereas the RfD_o is based on external (i.e., ingested) dose. Very little information is available concerning the toxicologic effectiveness of dermally-absorbed contaminants. In cases where sufficient data are available to determine DDI, the DDI can be transformed to an equivalent oral DI by multiplying the DDI by RE_D, or the ratio of the dermally absorbed dose to the orally absorbed dose as shown in Equation 15-1:

$$DI = DDI \cdot RE_D \qquad \text{(Equation 15-1)}$$

where:

DI = daily intake (oral equivalent) (mg/kg/day)
DDI = daily dermal intake (mg/kg/day)
RE_D = reciprocal of the absorption fraction for ingested chemical

The RE_D is the reciprocal of the gastrointestinal absorption fraction, which is equivalent to the RE for an absorbed dermal dose relative to an unabsorbed ingested dose, assuming that the absorbed dose is equivalently effective regardless of exposure route. The equivalent oral DI can then be compared with the RfD_o or used to determine the ER by the oral route. It is recognized that the assumption of equivalent effectiveness of absorbed doses between exposure routes does not hold in many cases, such as when effects occur at the portal of entry or when removal, inactivation or activation of the compound before reaching the target organ varies with the exposure route.

In some cases, potency estimates have been derived from a different route of exposure than that which may occur from food chain contamination (e.g., inhalation). In these cases, the use of RE for carcinogens is similar to that described for threshold-acting toxicants.

The RE factor should only be applied when well documented and referenced information about the pharmacokinetics of the contaminant are available. Frequently such experimental data are lacking and RE may be set equal to 1; that is, exposure routes are assumed to be equally effective. If the REs for adults and children by the same route of exposure are known to be significantly different, the DI for each exposure pathway can be modified by the route-specific RE prior to adjustment for exposure duration and calculation of total DI (see Sections 15.2.2. and 15.2.3.).

15.2.2. Exposure Duration Adjustment (EDA). Human risk assessments for nonthreshold (carcinogenic) toxicants can be determined for lifetime (70 years) exposure or for exposures of shorter duration (e.g., 16 or 30 years; see Section 2.4.4.). It is assumed that the individual is exposed throughout this period to a longterm average concentration of the pollutant in the environment (i.e., soil or water) that resulted from continual combustor emissions.

An exposure duration adjustment (EDA) may be used when exposure duration is shorter than the assumed 70-year lifetime. EDA, analogous to a time-weighted average in most cases, is the exposure duration divided by 70 years. For example, in Scenario A, an adult is exposed for 16 years and DI can be multiplied by an EDA of 16/70 years to estimate a lifetime average intake. EDA may also be used for exposures that may be more prevalent in children. For example, soil ingestion in children (i.e., pica) is most prevalent up to 5 years of age. Thus, DI from soil ingestion can be multiplied by an EDA of 5/70 years to estimate a lifetime average.

For some exposure scenarios, an EDA may be used to estimate childhood exposure and also exposure during part of adulthood. For example, in Scenario B described in this document, DI is determined for an individual during both childhood and adulthood. An EDA of 5/30 is applied to the DI of the child and 25/30 is multiplied by the intake of the adult to estimate total DI during the 30 year exposure period. To further adjust for lifetime exposure, the 30 year DI is multiplied by 30/70 years to estimate an average daily dose over the lifetime. Equation 15-2 shows how an exposure duration-adjusted daily intake can be determined.

$$DI_A = [(DI(child) \cdot EDA) + (DI(adult) \cdot EDA)] \cdot D_S/L_d \qquad \text{(Equation 15-2)}$$

where:

DI_A = daily intake-adjusted for pathway (mg/kg/day)
DI = daily intake for pathway (mg/kg/day)
EDA = exposure duration adjustment (unitless)
D_S = duration of scenario (yrs)
L_d = lifetime duration (assumed 70 yrs)

Exposure duration adjustment is used only with carcinogens and should be carefully evaluated on a case by case basis, since for some carcinogens, childhood may be the critical exposure period and the carcinogenic risk would be underestimated by averaging over the entire lifetime.

15.2.2.1. EXAMPLE CALCULATION OF DAILY INTAKE-ADJUSTED-- Daily intake of benzo(a)pyrene by each pathway in Scenario B is described in Chapters 5-7 and 9-12 and is shown in Table 15-1. Use of exposure duration adjustments for these pathways is illustrated by applying Equation 15-2 to the food ingestion pathway to give the daily intake-adjusted:

$$\begin{aligned} DI_A &= [(3.61\times10^{-5} \cdot 5/30) + (7.23\times10^{-6} \cdot 25/30)] \cdot 30/70 \\ &= 5.16\times10^{-6}\ \text{mg/kg/day} \end{aligned}$$

Table 15-2 shows the exposure duration-adjusted daily intake of benzo(a)pyrene for the various pathways. The DI and DI_A for dermal exposure are not determined since data an input variable were not available (refer to Chapter 7).

15.2.3. Estimating Total Exposure by Each Route. Total daily intake (TDI) can be estimated by summing the DI_A for all pathways of the same exposure route relevant to the exposure scenario. In this way, a TDI for oral (food, water, soil and fish) exposure and a TDI for dermal (soil and water) exposure are estimated. (TDI for inhalation exposure could also be included if incorporated into the exposure scenario.) A limitation in this approach is that potential interactions by multiple exposure routes are not evaluated. However, to determine the effective TDI, the DI (DI_A for carcinogens) for each exposure route must be divided by that route's relative effectiveness (RE) factor.

TABLE 15-1

Results of Example Calculations of Daily Intake of Benzo(a)pyrene in Scenario B[a] (mg/kg/day)

Pathway	Child	Adult
Food Ingestion	$3.61\text{x}10^{-5}$	$7.23\text{x}10^{-6}$
Dermal Absorption from Soil	NA[b]	NA
Soil Ingestion	$4.51\text{x}10^{-9}$	$4.38\text{x}10^{-11}$
Water Ingestion	$1.02\text{x}10^{-8}$	$3.72\text{x}10^{-9}$
Fish Ingestion	$1.93\text{x}10^{-9}$	$1.66\text{x}10^{-9}$

[a] Presented as examples only; see text
[b] NA = not available (refer to Chapter 7 for explanation)

TABLE 15-2

Results of Example Calculations of Daily Intake-Adjusted of Benzo(a)pyrene in Scenario B[a] (mg/kg/day)

Pathway	Intake
Food Ingestion	5.16×10^{-6}
Dermal Absorption from Soil	NA[b]
Soil Ingestion	3.39×10^{-10}
Water Ingestion	2.06×10^{-9}
Fish Ingestion	7.31×10^{-10}
All Pathways	5.16×10^{-6}

[a] Presented as example only; see text

[b] NA = not available (refer to Chapter 7 for explanation)

Thus, TDI is dependent on the DI (or DI_A) for each pollutant, h, and RE for each exposure pathway, k, and can be derived using the following equations:

$$TDI_{hk} = \frac{DI_{hkx}(food)}{RE(food)} + \frac{DI_{hkx}(water)}{RE(water)} + \frac{DI_{hkx}(soil)}{RE(soil)} + \frac{DI_{hkx}(fish)}{RE(food)}$$

(Equation 15-3)

where:

TDI_{hk} = total daily intake for the h^{th} pollutant by k^{th} route (mg/kg/day)
DI_{hk} = daily intake of h^{th} pollutant from a given exposure route/pathway, k (mg/kg/day)
RE = relative effectiveness of the route of exposure, with respect to the pertinent route of exposure (e.g., food) (unitless)
h denotes route, which is oral in this equation
A is substituted for x for carcinogens (DI_A should be used)

$$TDI_{hk} = \frac{DI_{hkx}(dermal\text{-}soil)}{RE(dermal\text{-}soil)} + \frac{DI_{hkx}(dermal\text{-}water)}{RE(dermal\text{-}water)}$$

(Equation 15-4)

where:

TDI_{hk} = total daily intake for the h^{th} pollutant by k^{th} route (mg/kg/day)
DI_{hk} = daily intake of h^{th} pollutant from a given exposure route/pathway, k (mg/kg/day)
RE = relative effectiveness of the route of exposure, with respect to the pertinent route of exposure (e.g., dermal) (unitless)
k denotes the route, which is dermal in this equation
A is substituted for x for carcinogens (DI_A should be used)

Since fish is a food group the relative effectiveness for food is used in the determination of a total daily oral intake.

15.2.4. Total Background Intake of Pollutant by Each Route (TBI). It is important to recognize that there may be sources of exposure other than the combustor emissions, and that total exposure should be maintained below the RfD or a specific risk level. These sources of exposure include background levels (whether natural or anthropogenic) in drinking water, food or air. Other types of exposure that are due to occupation or habits such as smoking might also be included, depending on data availability and regulatory policy. These exposures are summed to estimate TBI.

Data for estimating background exposure usually are obtained from analytical surveys of surface, ground or tap water, from FDA market basket surveys, and from air monitoring surveys. These surveys may report means, medians, percentiles or ranges, as well as detection limits. Estimates of TBI may be based on values representing central tendency or on upper-bound exposure situations, depending on the desired degree of conservatism.

Data chosen to estimate TBI should be consistent with the value of body weight. Where background data are reported in terms of a concentration in air or water, ingestion or inhalation rates applicable to adults or children can be used to estimate the daily background intake value. Where data are reported as total daily dietary intake for adults and similar values for children are unavailable, conversion to an intake for children may be required. Such a conversion could be estimated on the basis of relative total food intake or relative total caloric intake between adults and children. Background intake may also be adjusted for duration if such information is available for children and adults.

The TBI is the summed estimate of all possible background exposures except exposures resulting from combustor emissions. The effective TBI is determined by dividing the background intake values (BI) for the pollutant, h, for each exposure route, k, by the relative effectiveness (RE) factor of that route or source. Thus, after all the background exposures have been determined, TBI values can be calculated for oral and dermal exposure respectively, using the following equation:

$$TBI_{hk} = \frac{BI_{hk}(\text{food})}{RE(\text{food})} + \frac{BI_{hk}(\text{water})}{RE(\text{water})} + \frac{BI_{hk}(\text{soil})}{RE(\text{soil})} + \frac{BI_{hk}(\text{fish})}{RE(\text{food})}$$

(Equation 15-5)

where:

TBI_{hk} = total background intake rate of h^{th} pollutant by k^{th} route (mg/kg/day)
BI_{hk} = background intake of h^{th} pollutant from the k^{th} route (mg/kg/day)
RE = relative effectiveness, with respect to the pertinent route of exposure (e.g., food) (unitless)
k denotes route, which is oral in this equation

$$TBI_{hk} = \frac{BI_{hk}(\text{soil})}{RE(\text{soil})} + \frac{BI_{hk}(\text{water})}{RE(\text{water})}$$

(Equation 15-6)

where:

TBI_{hk} = total background intake of h^{th} pollutant by k^{th} route, (mg/kg/day)
BI_{hk} = background intake of h^{th} pollutant by k^{th} route (mg/kg/day)
RE = relative effectiveness, with respect to the pertinent route of exposure (unitless)
k denotes route, which is dermal in this equation

A limitation in determination of the TBI is that potential interactions (e.g., synergism or antagonism) by multiple exposure routes are not evaluated.

Total background intake should be derived on a site-specific basis. Lacking such data, background intake for the particular geographic area is preferred over national averages. The degree of urbanization where the exposed individual lives should also be considered since the contribution to exposure from ambient concentrations of certain pollutants (e.g., lead) may differ.

When TBI is subtracted from the RfD, the remainder (after adjusting for RE) defines the increment that can result from combustor emissions without exceeding the threshold. That is, the TDI plus the TBI should not exceed the RfD. If upper-bound data (such as 95th percentiles) were used to estimate TBI, then an increase in exposure corresponding to this increment, if realized, would cause the RfD to be approached or exceeded in a relatively small percentage (5%) of the exposed population. If central-tendency data (the median) were used to estimate TBI, such an increase would cause the RfD to be approached or exceeded in about half of the exposed population. If TBI were set at zero for lack of exposure data, the allowed increase would result in exceeding the RfD to an unknown degree, depending on whether other sources of exposure exist.

Total background intake is not considered for nonthreshold toxicants in this methodology since ER is determined and is an estimate of incremental risk and not total carcinogenic risk (see Section 15.3.3.).

15.3. RISK ESTIMATION

The scenarios presented in this document do not determine exposure or risk for an entire exposed population (aggregate risk). Rather, they determine the risk for an individual in a defined exposure scenario without reference to the numbers of individuals at a given exposure level. For estimation of risk to threshold-acting toxicants (i.e., noncarcinogens), DI by ingestion (food, water and/or soil) or dermal exposure (soil) is compared with the RfD to characterize the risk to human health. A DI exceeding the RfD would be a basis for concern that adverse health effects may occur in those individuals.

For carcinogens, the DI is used to estimate excess carcinogenic risk. The excess risk (ER) is defined as the incremental lifetime cancer risk above background to a hypothetical population in which all individuals are continuously exposed to a concentration equal to the daily intake of the contaminant. The ER is derived from the daily incremental dose of the contaminant above background and the human cancer potency factor (e.g., q_1^*) as established by U.S. EPA methods (U.S. EPA, 1986f). For example, an ER of 10^{-6} means that lifetime exposure to that specific concentration (or administered dose) would have an upper-bound excess cancer risk of one case in one million individuals.

Exposure to members of a chemical class or a chemical congener may be estimated from emission data using the proportion of the total mass that the congener comprises of the chemical class. Unless congener-specific data are available, all the congeners of the chemical class (e.g., PAHs or PCBs) are assumed to have the same transport and fate characteristics. Furthermore, for the purposes of risk assessment, the congener is assumed to comprise the same proportion of the total chemical class in deposition, soils, plants and animals as it does in the emissions. This allows estimation of exposure to the congener as well as to the total chemical class. Potential health effects resulting from exposure to the congener or the chemical class can be evaluated using the RfD or ER for the congener.

15.3.1. Comparison with the RfD. For threshold-acting toxicants, risks due to the DI from the various pathways in the exposure scenarios are assessed by comparison with the RfD. Values of DI from the oral route (ingestion of food, soil, water and fish) are summed to determine TDI_{hk} (where k is the oral route) and compared with the oral RfD. Pathways of exposure by dermal absorption of contaminants in soil (and water, if available) may be converted to an equivalent oral DI, summed and compared with the oral RfD.

One approach for assessment of risk to systemic toxicants by multiple routes of exposure is the use of an approach analogous to a hazard index (HI) (Stara et al., 1987), wherein the total daily intake for the pollutant (h) is expressed relative to the RfD for the pollutant by the same exposure route as shown in the following equation:

$$HI_{hk} = TDI_{hk}/RfD_{hk} \quad \text{(Equation 15-7)}$$

where:

HI_{hk} = hazard index for the h^{th} pollutant by the k^{th} exposure route (unitless)
TDI_{hk} = total daily intake for the h^{th} pollutant by the k^{th} route (such as inhalation, dermal) (mg/kg/day)
RfD_{hk} = reference dose for chronic exposure to the h^{th} pollutant by the k^{th} route (k is the same route as the TDI) (mg/kg/day)

If RfDs for several routes of exposure are available, a multi-route (total) hazard index for a chemical, h, can be determined according to the following equation:

$$HI_h = \sum_{k=1}^{m} TDI_{hk}/RfD_{hk} \quad \text{(Equation 15-8)}$$

where:

HI_h = hazard index for the h^{th} chemical for m exposure routes, k (unitless)
TDI_{hk} = total daily intake for h^{th} chemical by k^{th} route (mg/kg/day)
RfD_{hk} = reference dose for chronic exposure to h^{th} chemical by k^{th} route (where k is the same route as the TDI) (mg/kg/day)

It is assumed that no hazard exists if the HI_h is less than 1, since the TDI does not exceed the RfD. If the HI is greater than 1, the risk is defined only in terms relative to the RfD. This approach can only be used if the route-specific RfDs (i.e., RfD_o, RfD_i) are not based on portal-of-entry effects.

For multiple chemical exposures, such as with mixtures, by several routes, a hazard index for each chemical by multiple routes could be determined (HI_h). These values of HI_h may then be summed for groups of chemicals affecting a given target tissue according to Equation 15-9:

$$HI^{Mix} = \sum_{h=1}^{n} HI_h \qquad \text{(Equation 15-9)}$$

where:

HI^{Mix} = hazard index for a mixture of n chemicals (unitless)
HI_h = hazard index for multiple route exposure to h^{th} chemical (unitless)

The summing of HI_h for pollutants can only be used if the RfDs for those chemicals are based on effects in the same target organ.

15.3.1.1. EXAMPLE CALCULATION: COMPARISON WITH RfD FOR CADMIUM. The definition and derivation of the RfD was discussed in Chapter 14. The total background intake (TBI) of cadmium for adults has been estimated to be 27 μg/day (3.85×10^{-4} mg/kg/day) (Federal Register, 1985). The oral RfD for cadmium in food is 1×10^{-3} mg/kg/day and the oral RfD for cadmium in water is 5×10^{-4} mg/kg/day (U.S. EPA, 1985b; U.S. EPA, 1990). It is implied that children would not be adversely affected when exposed to the same food and water that resulted in this threshold level in adults. The DI of cadmium for the various pathways in Scenario B are shown in Table 15-3. Scenario B is described in Chapter 2, and DI values for cadmium were derived in example calculations presented in Chapters 3-12. Although based on current research, these calculations are provided only as examples and should not be taken as definitive determinations of cadmium exposure for any particular combustion facility.

TABLE 15-3

Results of Example Calculations of Daily Intake of Cadmium in Scenario B[a]

(mg/kg/day)

Pathway	Child	Adult
Food Ingestion	6.62×10^{-5}	1.80×10^{-5}
Dermal Absorption from Soil	NA[b]	NA
Soil Ingestion	2.39×10^{-5}	2.32×10^{-7}
Water Ingestion	1.18×10^{-5}	4.30×10^{-6}
Fish Ingestion	3.95×10^{-6}	3.41×10^{-6}

[a] Presented as examples only; see text

[b] NA = not available (refer to Chapter 7 for explanation)

Since an RfD for cadmium is available for both food and water ingestion, pathways involving exposure from the same sources would be compared with the respective RfD. DI from ingestion of food, soil and fish are summed and the TDI compared with the oral RfD for food, whereas the DI from water ingestion is compared with the oral RfD for water.

The TDI for cadmium for food ingestion and water ingestion routes is determined according to Equation 15-3. An RE of 1 relative to food is assumed, and adult values are used.

$$TDI_{food} = \frac{1.80x10^{-5}}{1} + \frac{2.32x10^{-7}}{1} + \frac{3.41x10^{-6}}{1}$$

$$= 2.16x10^{-5} \text{ mg/kg/day}$$

$$TDI_{water} = 4.30x10^{-6} \text{ mg/kg/day}$$

Since there is not a RfD for chronic dermal exposure, the DDI of cadmium in soil could be converted to an equivalent oral DI, summed with oral DIs and the TDI compared with one of the oral RfDs. However, data were not available for the absorption fraction of cadmium in soil for human skin and DDI was not calculated (see also Chapter 7).

Hazard indices are calculated for food ingestion and water ingestion since there are RfDs for these routes. The hazard indices are summed to determine overall risk relative to the RfD. The TBI for cadmium can also be subtracted from the RfD to show the increment that can result from combustor emissions without exceeding the threshold.

The HI for cadmium for food ingestion according to Equation 15-7 is:

$$\frac{2.16\text{x}10^{-5}}{1\text{x}10^{-3}} = 2.2\text{x}10^{-2}$$

The HI for cadmium for water ingestion is:

$$\frac{4.30\text{x}10^{-6}}{5\text{x}10^{-4}} = 8.6\text{x}10^{-3}$$

The HI for cadmium for these routes is:

$$\begin{aligned} HI &= 2.2\text{x}10^{-2} + 8.6\text{x}10^{-3} \\ &= 3.0\text{x}10^{-2} \end{aligned}$$

If there were several chemicals with the same target organ, HI_h could be determined for each chemical across its routes of exposure and HI determined for the mixture according to Equation 15-9. Since for cadmium HI <1, the appropriate conclusion is that cadmium does not present a risk of noncarcinogenic effects by Scenario B. As stated in Section 13.3.1., cadmium is classified as a probable human carcinogen, but insufficient information exists to derive a quantitative estimate of carcinogenic risk for ingested cadmium.

15.3.2. Determination of the Excess Risk Level (ER) for Non-Threshold Toxicants (Carcinogens). For chemicals classified as carcinogens, the excess risk, ER, for a particular route of exposure is determined as follows:

$$ER_{hk} = 1\text{-}\exp[-(TDI_{hk} \cdot RE \cdot B_{Hhk})]$$

(Equation 15-10)

where:

ER_{hk} = excess risk for the h^{th} pollutant by k^{th} exposure route (unitless)
TDI_{hk} = total daily intake for the h^{th} pollutant by the k^{th} route (mg/kg/day)
RE = relative effectiveness, with respect to pertinent route of exposure (unitless)
B_{Hhk} = human cancer potency (B_H) for the h^{th} pollutant by k^{th} route [per(mg/kg)/day]

The definition and derivation of each of the parameters used to estimate ER for carcinogens have been discussed in Sections 15.2.1., 15.2.2. and 15.2.4. and Chapter 14. TBI is not taken into account in the above equation, since ER is an expression of incremental risk due to the facility and not an expression of total carcinogenic risk. It may be useful, however, to compare TBI with DI_A or TDI to determine the fraction of exposure to a given carcinogen that is contributed by the facility.

ER is determined for each exposure route if a human cancer potency estimate is available for that exposure route. Risk across routes can be summed to determine the multiroute exposure risk for chemical i by the following equation:

$$ER_h = \sum_{k=1}^{m} ER_{hk}$$

(Equation 15-11)

where:

ER_h = excess risk for h^{th} chemical for m exposure routes (unitless)
ER_{hk} = excess risk for h^{th} chemical for k^{th} exposure route (unitless)

This equation assumes independence of action for each route and is equivalent to the assumption of route addition (dose and response addition across routes). A hazard index approach using a specific level of risk could also be applied, somewhat analogous to that for systemic toxicants (U.S. EPA, 1986g).

For a mixture of carcinogenic chemicals, excess risk can be estimated by the following equation (Stara et al., 1987):

$$ER^{Mix} = 1 - \prod_{h=1}^{n} (1 - ER_h) \qquad \text{(Equation 15-12)}$$

where:

ER^{Mix} = excess risk for a mixture of n chemicals for multiroute exposure (unitless)

ER_h = excess risk for the h^{th} chemical for m exposure routes (unitless)

This approach assumes independence of action by the carcinogens in a mixture. Because of uncertainties in estimating dose-response relationships for both single and multiple chemicals, the excess risk should not be regarded as an estimate of absolute risk. The assumptions, limitations and uncertainties in conducting risk assessment of mixtures are detailed in U.S. EPA (1986g).

15.3.2.1. EXAMPLE CALCULATION OF ER FOR BENZO(A)PYRENE.

Excess risk calculations for each pathway of Scenario B are shown below. The human cancer potency for oral exposure (q_1*) for benzo(a)pyrene (B(a)P) has been calculated by the U.S. EPA to be 11.5 per(mg/kg)/day (U.S. EPA, 1980b). For purposes of illustration, it will be assumed that this potency value is the same for all routes of exposure. The TBI (i.e., intake from background sources of benzo(a)pyrene exposure not related to the combustor) for adults is estimated to be ≈0.88 μg/day (1.26×10^{-3} mg/kg/day) (U.S. EPA, 1980b).

Risk due to TBI is calculated using Equation 15-10:

$$\begin{aligned}\text{Risk} &= 1 - \exp[-(1.26\times10^{-3} \cdot 1 \cdot 11.5)] \\ &= 1.44\times10^{-2}\end{aligned}$$

The DI_A for benzo(a)pyrene has been calculated for each pathway in Scenario B (see Table 15-2). Scenario B is described in Chapter 2, and DI values for benzo(a)pyrene were derived in example calculations presented in Chapters 3-12. Although bases on current research, these calculations are provided only as examples and should not be taken as definitive determinations of benzo(a)pyrene exposure for any particular combustion facility. Values of ER for each pathway are calculated from the respective DI_A as follows:

For the Food Ingestion Pathway:

$$\begin{aligned}ER &= 1-\exp[-(5.16\times10^{-6} \cdot 1 \cdot 11.5)] \\ &= 5.93\times10^{-5}\end{aligned}$$

For the Dermal Absorption from Soil Pathway:

ER = Cannot be quantitated as input data are not available for calculation of DI

For the Soil Ingestion Pathway:

$$\begin{aligned}ER &= 1-\exp[-(3.39\times10^{-10} \cdot 1 \cdot 11.5)] \\ &= 3.90\times10^{-9}\end{aligned}$$

For the Water Ingestion Pathway:

$$ER = 1-\exp[-(2.06\times10^{-9} \cdot 1 \cdot 11.5)]$$
$$= 2.37\times10^{-8}$$

For the Fish Ingestion Pathway:

$$ER = 1-\exp[-(7.31\times10^{-10} \cdot 1 \cdot 11.5)]$$
$$= 8.41\times10^{-9}$$

The excess risk for benzo(a)pyrene all routes of exposure is:

$$ER = 5.93\times10^{-5} + 3.90\times10^{-9} + 2.37\times10^{-8} + 8.41\times10^{-9}$$
$$= 5.93\times10^{-5}$$

These values represent a plausible upper-bound for excess risk resulting from Scenario B. It can be seen that risk due to TBI (i.e., that due to background sources of benzo(a)pyrene not related to the combustor) is much greater than the calculated excess risk (i.e., that due to the combustion facility).

15.4. CHARACTERIZATION OF UNCERTAINTY

There is an inherent uncertainty in the models presented in this document to determine exposure to combustor emissions. An uncertainty analysis can be used to estimate a range of model values from which a best estimate can be derived. Such an uncertainty analysis was undertaken for the terrestrial food chain and human soil ingestion pathways by Belcher and Travis (1989). Pollutant deposition rates were modeled for 24 operating MWCs in the United States. These rates and other input variables were expressed as distributions and were sampled using a stratified, random sampling procedure referred to as Latin Hypercube Sampling.

A range of contaminant intake values was determined from many iterations of the procedure. Because of differences between pollutant deposition rates used in Scenario B of the present analysis and those included in the Belcher and Travis (1989) analysis, the results of the two analyses in terms of pollutant intake are not directly comparable. Details of the latter analysis will not be presented here. However, the results for cadmium and benzo(a)pyrene, presented below, serve to illustrate the potential variability in computed pollutant intakes that can result from variation of input parameters.

The Belcher and Travis (1989) analysis also examined the percentage contribution of each component of the studied pathways to total intake for each contaminant. The resulting breakdowns for cadmium and benzo(a)pyrene are also presented below.

15.4.1. Uncertainty Analysis for Cadmium. Cadmium intake from the terrestrial food chain and soil ingestion pathways was due primarily to root uptake by vegetation (63% of total daily intake) and consumption of contaminated meat and dairy products (21%). Tables 15-4, 15-5 and 15-6 show these contributions. A probability distribution of daily human intake by these pathways found a geometric mean of 2.1×10^{-3} µg/day (Figure 15-1), with a range of uncertainty (from approximately the 2nd to the 98th percentile of frequency) spanning approximately 2.7 orders of magnitude.

15.4.2. Uncertainty Analysis for Benzo(a)pyrene. Human consumption of contaminated vegetation was the primary source of human exposure to benzo(a)pyrene, contributing 72% of the total for the pathways analyzed. The contribution of various vegetative and meat groups to DI were also analyzed. Garden fruits were found to make the greatest contribution (33%) of vegetative groups and beef the greatest contribution (14%) for meat and dairy products. Results are shown in Tables 15-7, 15-8 and 15-9. Figure 15-2 shows the probability distribution for the daily intake of benzo(a)pyrene. The range of uncertainty was approximately 4 orders of magnitude, or somewhat greater than for cadmium.

A similar approach to an uncertainty analysis can also be conducted for other chemicals and the other pathways in this methodology.

15.5. CONCLUSION

The risk due to indirect pathways of exposure to combustor emissions is characterized differently for carcinogens and noncarcinogens. Risk of carcinogens is determined by calculation of an upper-bound estimate of excess carcinogenic risk. Risk of noncarcinogens is calculated by comparison with an RfD using a hazard index approach. Both approaches can be applied to the same

TABLE 15-4

Breakdown of Cadmium Exposure for the Terrestrial Food Chain and Human Soil Ingestion Pathways (% contributions to average daily intake)*

1.	Direct ingestion of contaminated soil:	12.9%
2.	Consumption of contaminated vegetation:	
	Atmospheric deposition component	3.3%
	Root uptake component	62.8%
	Air-to-plant transfer component	0.0%
3.	Consumption of contaminated meat and dairy products:	
	Soil-Animal-Human Route	13.3%
	Plant-Animal-Human Route	7.7%

*Source: Belcher and Travis, 1989

TABLE 15-5

Contribution to Daily Intake of Cadmium by Vegetation Food Groups*

Source	Percentage of the Total Intake
Garden fruits	32.3%
Root vegetables	19.6%
Potatoes	6.9%
Leafy vegetables	4.7%
Dried legumes	1.7%
Grains	0.6%
Fresh legumes	0.3%

*Source: Belcher and Travis, 1989

TABLE 15-6

Contribution to Daily Intake of Cadmium by Meat Food Groups[*]

Source	Percentage of the Total Intake
Beef liver	12.3%
Dairy products	3.7%
Poultry	3.1%
Beef	1.4%
Pork	0.3%
Eggs	0.2%
Lamb	<0.1%

[*] Source: Belcher and Travis, 1989

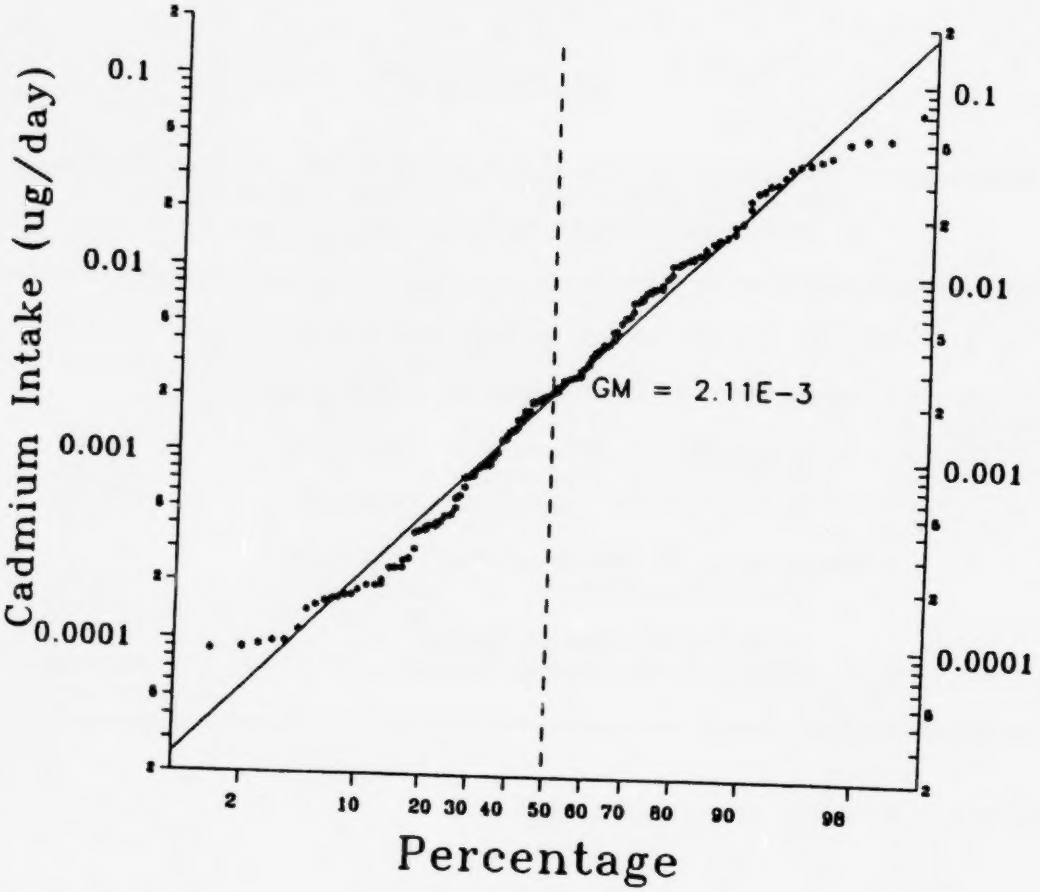

Figure 15-1. Probability distribution for cadmium daily intake values by the Terrestrial Food Chain and Human Soil Ingestion pathways

Source: Belcher and Travis, 1989

TABLE 15-7

Breakdown of Benzo(a)pyrene Exposure for the Terrestrial Food Chain and Human Soil Ingestion Pathways (% contributions to average daily intake)[*]

1.	Direct ingestion of contaminated soil:	4.3%
2.	Consumption of contaminated vegetation:	
	Atmospheric deposition component	7.2%
	Root uptake component	15.0%
	Air-to-plant transfer component	50.2%
3.	Consumption of contaminated meat and dairy products:	
	Soil-Animal-Human Route	3.3%
	Plant-Animal-Human Route	20.0%

[*] Source: Belcher and Travis, 1989

TABLE 15-8

Contribution to Daily Intake of Benzo(a)Pyrene by Vegetation Food Groups*

Source	Percentage of the Total Intake
Garden fruits	32.9%
Leafy vegetables	12.8%
Potatoes	12.1%
Dried legumes	11.0%
Fresh legumes	2.6%
Root vegetables	0.8%
Grains	<0.1%

*Source: Belcher and Travis, 1989

TABLE 15-9

Contribution to Daily Intake of Benzo(a)Pyrene by Meat Food Groups*

Source	Percentage of the Total Intake
Beef	13.7%
Dairy products	5.8%
Beef liver	3.5%
Lamb	0.2%
Pork	0.1%
Poultry	<0.1%
Eggs	<0.1%

*Source: Belcher and Travis, 1989

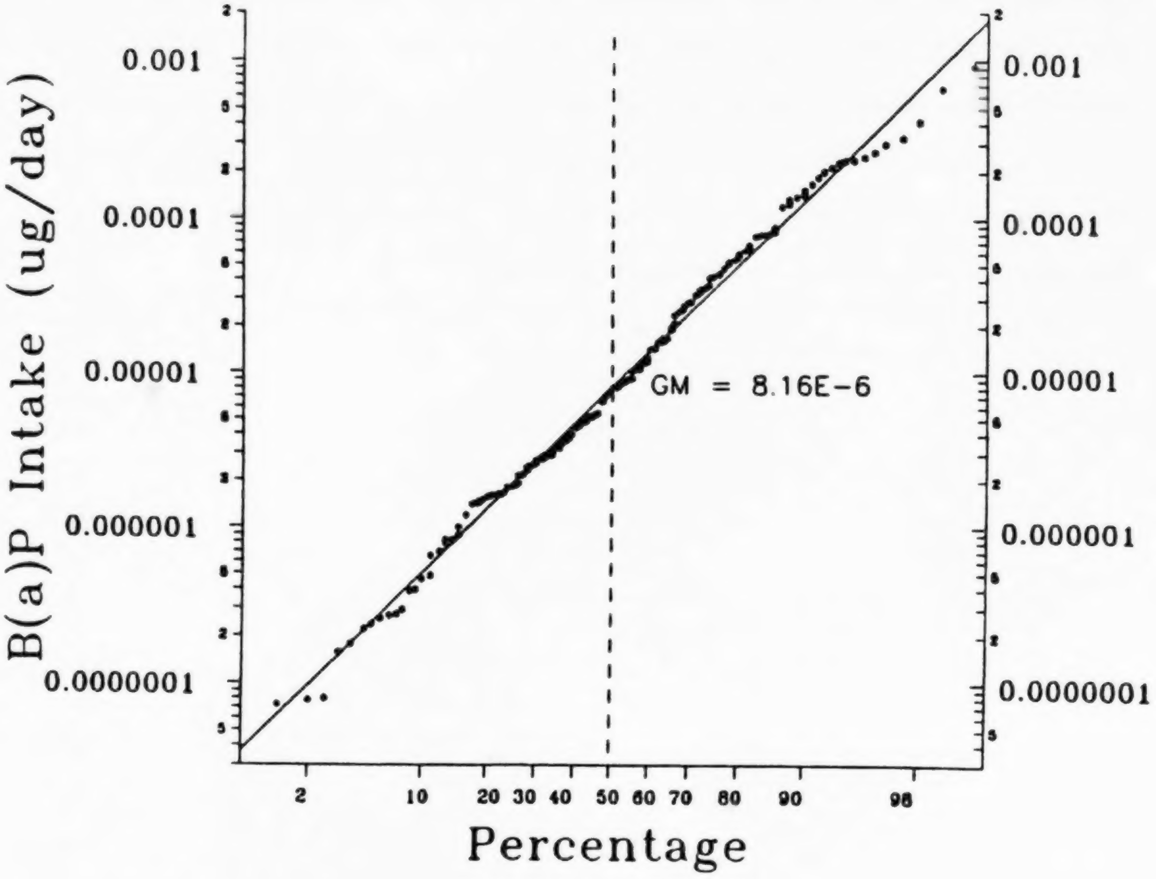

Figure 15-2. Probability distribution for benzo(a)-pyrene daily intake values for the Terrestrial Food Chain and Human Soil Ingestion pathways

Source: Belcher and Travis, 1989

chemical across multiple exposure routes. These approaches can also be applied to multiple chemicals in a mixture across multiple exposure routes, with some constraints.

Uncertainty analysis, such as may be carried out by a stochastic sampling approach, is a useful tool for examining the influence of particular assumptions, input values or pathways on the overall results, and for estimating overall ranges of uncertainty associated with the use of the methodology. Analyses of this type are particularly helpful whenever the interpretation and use of results derived from this methodology are important components of a risk management process.

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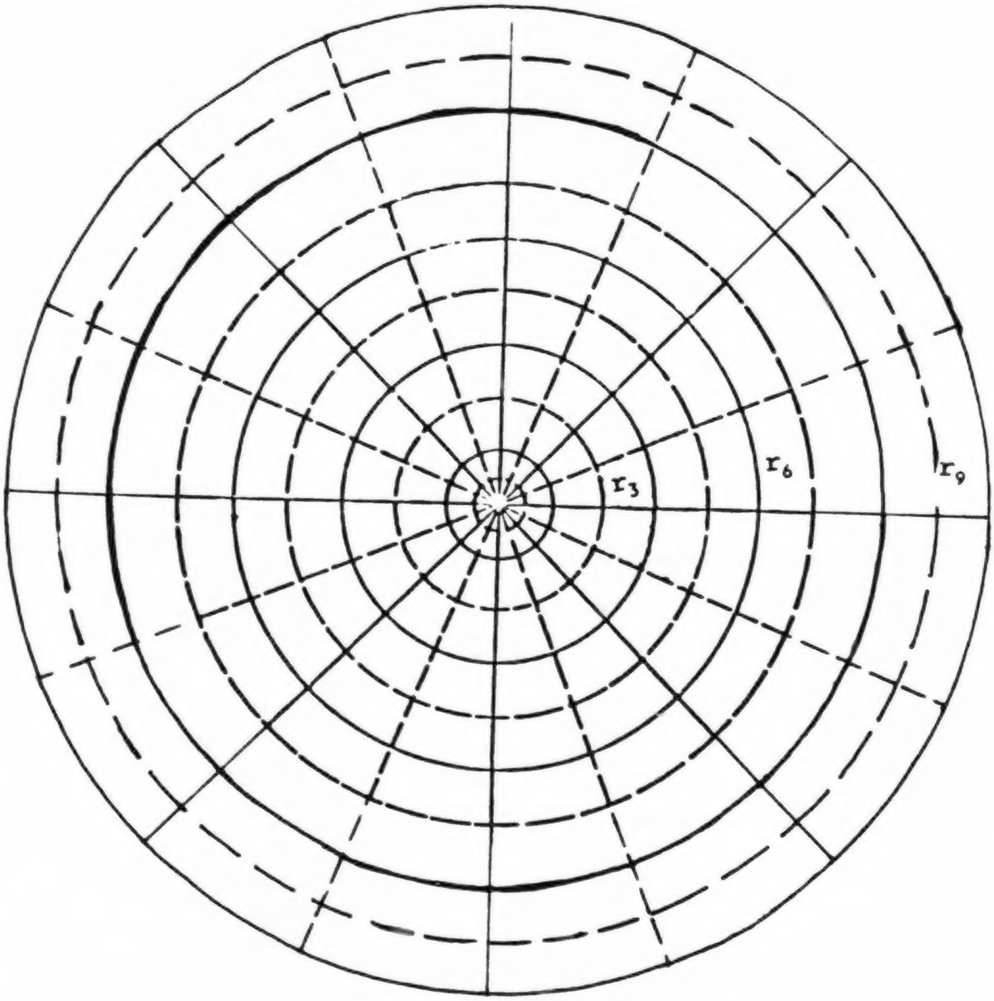
APPENDIX A



r = radius from source (m)

FIGURE A-1

The concentric rings used in the COMPDEP model.
Radii not drawn to scale.

TABLE A-1

Location of Receptor

Receptor Number	Radius (m)
1 - 16	200
17 - 32	500
33 - 48	1000
49 - 64	2000
65 - 80	5000
81 - 96	10000
97 - 112	20000
113 - 128	30000
129 - 144	40000
145 - 160	50000

APPENDIX A

Equation to Calculate Areal Average Deposition

$$Daa = \Sigma \left[\left(\frac{r_1 + r_2}{2}\right)^2 (D_1 - D_2) + \left(\frac{r_2 + r_3}{2}\right)^2 (D_2 - D_3) + \ldots\ldots\ldots + \left(\frac{r_{n-1} + r_n}{2}\right)^2 (D_{n-1} - D_n) / r^2 \right] + D_n$$

where:

Daa = areal average deposition ($g/m^2/yr$)
D = concentration at i^{th} radius
r = radius from source (m)

r_1 = 200 m
r_2 = 500 m
r_3 = 1000 m
r_4 = 2000 m
r_5 = 5000 m
r_6 = 10,000 m
r_7 = 20,000 m
r_8 = 30,000 m
r_9 = 40,000 m
r_{10} = 50,000 m